THE IMPAIRMENT OF BENEFICIAL USES IN LAKE ONTARIO

Final Report
submitted to the

GREAT LAKES ENVIRONMENT OFFICE
ENVIRONMENT CANADA

by

SARAH RANG, UNIVERSITY OF TORONTO;
JOHN HOLMES, UNIVERSITY OF TORONTO;
SANDY SLOTA, McMASTER UNIVERSITY;
DOUGLAS BRYANT, McMASTER UNIVERSITY;
EVERT NIEBOER, McMASTER UNIVERSITY; and
HENRY REGIER, UNIVERSITY OF TORONTO.

FEBRUARY 28, 1992

CONTRACT NUMBER KA401-0-0700/01-XSE

TABLE OF CONTENTS

AUTHORS' BIOGRAPHIES

DOUGLAS W. BRYANT received an undergraduate degree from McGill, and graduate degrees from York, followed by postdoctoral study at Oxford. Since 1978, Douglas has been at McMaster in the Department of Biochemistry. His major interests are in DNA repair, chemical mutagenesis and carcinogenesis, with a focus on the identification and characterization of polycyclic aromatic compounds (PAC) associated with urban air respirable particulates. He correlates specific chemical mutagens with unique mutational changes at the DNA level in bacterial and mammalian test systems. Such mutation "fingerprinting" focuses attention of analytical chemistry and biochemistry on compounds most likely to effect human health.

JOHN A. HOLMES received his undergraduate training at Guelph with his graduate work at Toronto, as a collaborative student at the Institute for Environmental Studies and Zoology. His M.Sc. work focussed on fisheries rehabilitation in the Hamilton Harbour-Cootes Paradise ecosystem. He is completing his Ph.D. studies in Zoology on the thermal ecology of larval sea lamprey and a first-order assessment of the potential impacts of future climate warming on sea lamprey populations in the Great Lakes. Through JAH Environmental he has assisted LGL Limited in an assessment for Ontario Hydro of the effects of the Pickering B Nuclear Generating Station on the Lake Ontario aquatic environment.

EVERT NIEBOER received his Master's degree in Chemistry at McMaster and his Ph.D. at Waterloo. He joined the McMaster Biochemistry Department and Occupational Health Program as Professor of Toxicology in 1980, after a decade as an Environmental Scientist at Laurentian University, Sudbury, Ontario. His research interests focus on the biochemistry, clinical/analytical chemistry and toxicology of metals, particularly cadmium, lead and nickel. He has developed methods, including indicator proteins, DNA lesions and metabolites in individuals exposed to metal contamination, to support environmental and occupational health assessments. He has also estimated numerical health risks for government and industrial agencies.

SARAH RANG attended the University of Toronto, Institute for Environmental Studies and focussed her M.Sc. on water quality in the Niagara River. At the Conservation Council of Ontario, Sarah was responsible for coordinating urban streams rehabilitation. As Executive Assistant to the Minister of Environment in 1987-90, she helped develop and implement new environmental legislation, government policy and programs including the Municipal – Industrial Strategy for Abatement, the Blue Box program and the CFC phase-out strategy. As Vice-President of Environmental Economics International, her area of expertise includes toxic contaminants in the Great Lakes, waste management and government environmental policy.

HENRY REGIER is a graduate of Queen's and Cornell. He joined Zoology of the University of Toronto in 1966 and became Director of the Institute for Environmental Studies in 1989. He has been a Commissioner for the Great Lakes Fishery Commission and a member of the Executive Committee of the Great Lakes Science Advisory Board of the International Joint Commission. His expertise is in the ecology of fish and freshwaters, especially with respect to the rehabilitation of degraded ecosystems. He has helped to create interjurisdictional governance processes and structures for the Great Lakes Basin.

SANDY SLOTA received her BSc (Biology) and diploma in Occupational Health and Safety, both from McMaster University. She is currently completing the Occupational Hygiene program at Mohawk College as well as the Master of Business Administration program at Niagara University.

ACKNOWLEDGEMENTS

The Study Team would like to acknowledge the efforts and contributions of the many people who contributed to the report. Griff Sherbin, Mike Goffin and Ron Shimizu helped to get the project off the ground and provided valuable advice. Darrell Piekarz and then Melanie Neilson were generous with their time and suggestions to guide the study to completion. Assistance with the literature retrieval and critical appraisal was provided by Nathalie LaViolette.

The study team interviewed over 100 people who supplied important data, advice and contacts. Gerry Mikol and Charles Zafonte provided suggestions on sources of information relating to the south shore of Lake Ontario. In addition, over 30 people attended a workshop to review the interim results of the project and their ideas and discussion have been incorporated into the final report. Mike Gilbertson also provided us with the benefit of his extensive experience in herring gull contaminants and cause-effect linkages. The report is a result of the help and assistance of these people and others who continue to contribute to the scientific knowledge and management of the Great Lakes.

PREFACE

Since 1987, Environment Canada, the U.S. Environmental Protection Agency, the Ontario Ministry of the Environment and the New York State Department of Environmental Conservation have been Parties to the Lake Ontario Toxics Management Plan. The goal of the Plan is a lake ecosystem that provides drinking water and fish that are safe for unlimited human consumption, and that allows natural reproduction of the most sensitive species such as the bald eagle, osprey, mink and river otter. Activities to reduce toxic inputs in order to achieve this goal are undertaken concurrently whenever possible.

Under the Canada-U.S. Great Lakes Water Quality Agreement, Environment Canada and the U.S. EPA are also committed to developing a Lakewide Management Plan (LAMP) for each of the Great Lakes. The Secretariat to the Lake Ontario Toxics Management Plan has undertaken to expand the Plan into a LAMP.

Lakewide Management Plans embody an ecosystem approach to restoring and maintaining beneficial uses of lake waters. As a first step towards a comprehensive identification of impaired beneficial uses and associated critical pollutants for Canadian waters, Environment Canada contracted the Institute for Environment and Health (University of Toronto and McMaster University) to produce this report. If a comparable document will be produced in the U.S.A. for the American side of Lake Ontario, then these two documents may be evaluated by the Lake Ontario Secretariat in order to designate a list of critical pollutants for Lake Ontario.

This document does not address the loss of fish and wildlife habitat in Lake Ontario. The impairment of this beneficial use is being assessed by the Lake Ontario Secretariat through a separate undertaking.

EXECUTIVE SUMMARY

Lake Ontario is a contaminated ecosystem. Persistent toxic chemicals are present in the water, sediment, benthos, phytoplankton, zooplankton, fish and humans in the Lake Ontario basin. While the overall concentrations of contaminants are lower than in the 1960s and 1970s, current concentrations are still causing impairments of beneficial uses. This study has employed a critical appraisal approach to relevant documents to identify locations in which impairments of beneficial uses occur and the causal pollutants associated with each impairment of beneficial use.

The data base for Lake Ontario is discrete, personalized and much of it is unpublished. While most scientists are willing to share data, it requires an intensive effort to discover who is the relevant expert and what data may be available. Often these data are in raw form; analysis and interpretation are difficult and time consuming. A related barrier to the identification of impairments is the lack of standardized methods for sampling, storage and analysis; this complicates the inter-comparision of data sets obtained by different researchers or at different times.

Three impairments – restrictions on fish and wildlife consumption, restrictions on dredging activities, and beach closures – occur at more than 10 locations in Lake Ontario. One of the impairments – tainting of fish and wildlife flavour – has not been documented in Lake Ontario. However, there is a wide discrepancy in the quantity and quality of the information available for different impairments. For example, there are few reports on benthic community structure in Lake Ontario, whereas there is an annual book published on fish consumption restrictions based on thousands of hours of work collecting, analysing and

reporting by multiple agencies. To some extent this reflects the historical interest of agencies, the perceived "human" importance of the impairment and legally driven requirements to measure contaminants. However, that the amounts of information vary needs to be taken into account when concluding that certain impairments occur more frequently than others.

The locations with the highest number of impairments are mainly the Areas of Concern: Niagara River, Hamilton Harbour, Toronto and the Bay of Quinte. In part, this reflects the greater information available from these areas based on intensive studies. The list of impairments is currently being evaluated for the Areas of Concern on the southern shore. There is a lesser degree of scientific research activity in the open lake waters and in locations which are not designated Areas of Concern.

Impairments of beneficial uses fall into two general categories: those based on ecological considerations such as community structure and those based on exceedence of a jurisdictional guideline, standard or criterion. It was easier to find evidence to support the existence of an impairment based on jurisdictional standards than on ecological factors, because the exceedence of a guideline itself constituted evidence that impairment was occurring. Therefore, most of the causal pollutants are determined by association with exceedence of jurisdictional standards. Some of the guidelines, standards and criteria used for assessing exceedences are based on heavy metals and conventional pollutants and do not include a wide range of organic contaminants. If there is no standard or guideline for an organic contaminant any impairment must be inferred from ecological considerations, for which the relevant information may be weak. For example, until recently (1991) there were no dredging-related guidelines for organic pesticides such as chlordane, aldrin and dieldrin.

Despite the fact that elevated concentrations of these compounds can be found in Lake Ontario sediments, they are not considered in the assessment of impairments based on restrictions on dredging activities.

Many of the papers identified contaminant concentrations without reference to an effect or vice-versa. This type of information was used to establish linkages between causes and effects using a weight-of-evidence approach. Gilbertson et al. (1991) used the framework of criteria derived from epidemiology to analyse information; theirs is an important subset of the criteria employed in our study.

Key criteria used to select causal pollutants in our study were presence of the pollutant in Lake Ontario at an elevated concentration and association of the pollutant with an impairment of beneficial use. Some of the contaminants identified are surrogates for chemical contamination rather than a contaminant by themselves. Some of the causal pollutants, such as PCBs and PAHs, were associated with more than one impairment. This indicates the ability of the contaminant to cause multiple effects on the ecosystem. Twenty-two causal pollutants/agents and surrogates were identified as associated with use impairment in Lake Ontario.

LIST OF FIGURES

LIST OF TABLES

LIST OF APPENDICES

CHAPTER 1 INTRODUCTION

1.1 THE GREAT LAKES WATER QUALITY AGREEMENT

The Great Lakes Water Quality Agreement (GLWQA), based on the Boundary Waters Treaty of 1909, has established binational scientific, legislative and policy directions on water quality in the Great Lakes basin. The 1972 Agreement has been amended several times, notably in 1978 and 1987. The 1987 amendments included: the addition of specific objectives to Annex 1; Annex 2 – Remedial Action Plans and Lakewide Management Plans; Annex 13 – Pollution from Non-point Sources; Annex 14 – Contaminated Sediment; Annex 16 – Pollution from Contaminated Groundwater; and Annex 17– Research and Development. Work under these amendments is carried out by the Parties to the Agreement, together with the local, provincial and state governments.

Annex 2, on Remedial Action Plans and Lakewide Management Plans, forms the framework for our study. This Annex outlines the general principles for developing Remedial Action Plans (RAPs) and Lakewide Management plans (LAMPs) such that each "shall embody a systematic and comprehensive ecosystem approach to restoring beneficial uses in Areas of Concern or in open waters .... They are to serve as important steps toward the virtual elimination of persistent toxic substances and towards restoring and maintaining the biological integrity of the Great Lakes Basin Ecosystem."

Eight of the 43 Areas of Concern in the Great Lakes basin are located on Lake Ontario: Niagara River, Hamilton Harbour, Metro Toronto, Port Hope, Bay of Quinte, Oswego River, Eighteen Mile Creek and Rochester. In each of these Areas of Concern, a RAP is being prepared. Each RAP will include the following elements:

- a definition and description of the environmental problems;
- a definition of the causes of the use impairment;
- an evaluation of remedial measures already in place;
- an evaluation of additional remedial measures to restore beneficial uses and a schedule for their implementation;
- an identification of the persons or agencies responsible for implementation;
- a process for evaluating remedial measures; and
- a description of surveillance and monitoring processes to track the effectiveness of remedial measures.

Each draft RAP is submitted to the International Joint Commission (IJC) for review.

Each of the RAPs on Lake Ontario are in different stages of completion, with the Hamilton Harbour and Bay of Quinte RAPs being the most advanced. Two RAPs are being prepared for the Niagara River, one by Canadian agencies and one by U.S. agencies.

Lakewide Management Plans are to the open water what RAPs are to the nearshore areas. LAMPs shall be designed to reduce the loadings of Critical Pollutants in order to restore beneficial uses. LAMPs shall not allow increases in pollutant loadings where specific objectives are not exceeded. Based on Annex 2 of the GLWQA, such plans shall include:

(i) A definition of the threat to human health or aquatic life posed by the Critical Pollutants, singly or in combination with another substance, including their contribution to the impairment of beneficial uses;

(ii) An evaluation of information on concentrations, sources and pathways of the Critical Pollutants in the Great Lakes system, including all information on loadings of the

Critical Pollutants from all sources, and an estimation of total loadings of the Critical Pollutants by modelling or other identified methods;

(iii) Steps to be taken pursuant to Article VI of this Agreement to develop the information necessary to determine the schedule of load reductions of Critical Pollutants that would result in meeting agreement objectives, including steps to develop the necessary standard approaches and agreed procedures;

(iv) A determination of the load reductions of Critical Pollutants necessary to meet Agreement objectives;

(v) An evaluation of remedial measures presently in place, and alternative additional measures that could be applied to reduce loadings of Critical Pollutants;

(vi) Identification of the additional remedial measures that are needed to achieve the reduction of loadings and to eliminate the contribution to impairment of beneficial uses from Critical Pollutants, including an implementation schedule;

(vii) Identification of the persons or agencies responsible for implementation of the remedial measures in question;

(viii) A process for evaluating remedial measure implementation and effectiveness;

(ix) A description of surveillance and monitoring to track the effectiveness of the remedial measures and the eventual elimination of the contribution to impairments of beneficial uses from the Critical Pollutants; and

(x) A process for recognising the absence of a critical pollutant in open lake waters."

Although Lakewide Management Plans will be prepared for each of the Great Lakes, current efforts focus on Lake Ontario and Lake Michigan. Work toward the Lake Ontario LAMP in effect originated with the Niagara River Toxics Management Plan and the Lake Ontario Toxics Management Plan, both begun before 1987. The most recent revision of the Lake Ontario Toxics Management Plan was presented in November 1990. Discussions are also underway to determine how to translate the work done in the Lake Ontario Toxics Management Plan to the formal Lake Ontario Lakewide Management Plan.

Annex 2 specifies 14 ways in which beneficial uses of the Lakes can be impaired (Table 1.1). These impairments are broadly based, extending from specific toxicological concerns such as fish tumors to more general social concerns such as aesthetics and economic impacts of added cost to agriculture and industry. The International Joint Commission has recently issued guidelines for listing and delisting of Areas of Concern based on these impairments (IJC, 1991), see Table 1.2. Critical pollutants, according to Annex 2 of the GLWQA, are "substances that persist at levels that, singly or in combination, are causing or are likely to cause, impairment of these beneficial uses despite past application of regulatory controls due to their:

(i) Presence in open lake waters;

(ii) Ability to cause or contribute to a failure to meet Agreement Objectives through their recognised threat to human health and aquatic life; or

(iii) Ability to bioacummulate."

Table 1.1. The fourteen impairments of beneficial uses as described in the Great Lakes Water Quality Agreement (IJC, 1989).

Impairments of beneficial uses as defined in Annex 2 means a change in the chemical, physical or biological integrity of the Great Lakes Water Quality system sufficient to cause any of the following:

(i) Restrictions on fish and wildlife consumption
(ii) Tainting of fish and wildlife flavour
(iii) Degraded fish and wildlife populations
(iv) Fish tumors or other deformities
(v) Bird or animal deformities or reproductive problems
(vi) Degradation of benthos
(vii) Restrictions on dredging activities
(viii) Eutrophication or undesirable algae
(ix) Restrictions on drinking water consumption or taste or odour problems
(x) Beach closings
(xi) Degradation of aesthetics
(xii) Added cost to agriculture or industry
(xiii) Degradation of phytoplankton and zooplankton populations
(xiv) Loss of fish and wildlife habitat

The documentation and identification of causal pollutants are early steps in activating an effective LAMP. It is now timely to document the impairment of beneficial uses in Lake Ontario as a prerequisite to the identification of causal pollutants in Lake Ontario. Recognizing that the Parties will need to reach a consensus on the final list of pollutants in Lake Ontario, this study has adopted the term "causal pollutants", which the Parties may use as a basis for formulating a Critical Pollutant list.

Table 1.2. Guidelines for recommending the listing and delisting of Great Lakes Areas of Concern.

USE IMPAIRMENT	LISTING GUIDELINE	DELISTING GUIDELINE
RESTRICTIONS ON FISH AND WILDLIFE CONSUMPTION	When contaminant levels in fish or wildlife populations exceed current standards, objectives or guidelines, or public health advisories are in effect for human consumption of fish or wildlife. Contaminant levels in fish and wildlife must be due to contaminant input from the watershed.	When contaminant levels in fish and wildlife populations do not exceed current standards, objectives or guidelines, and no public health advisories are in effect for human consumption of fish or wildlife. Contaminant levels in fish and wildlife must be due to contaminant input from the watershed.
TAINTING OF FISH AND WILDLIFE FLAVOUR	When ambient water quality standards, objectives, or guidelines, for the anthropogenic substance(s) known to cause tainting, are being exceeded or survey results have identified tainting of fish or wildlife flavor.	When survey results confirm no tainting of fish or wildlife flavor.
DEGRADED FISH AND WILDLIFE POPULATIONS	When fish and wildlife management programs have identified degraded fish or wildlife populations due to a cause within the watershed. In addition, this use will be considered impaired when relevant, field-validated, fish or wildlife bioassays with appropriate quality assurance/quality controls confirm significant toxicity from water column or sediment contaminants.	When environmental conditions support healthy, self-sustaining communities of desired fish and wildlife at predetermined levels of abundance that would be expected from the amount and quality of suitable physical, chemical and biological habitat present. An effort must be made to ensure that fish and wildlife objectives for Areas of Concern are consistent with Great Lakes ecosystem objectives and Great Lakes Fishery Commission fish community goals. Further, in the absence of community structure data, this use will be considered restored when fish and wildlife bioassays confirm no significant toxicity from water column or sediment contaminants.
FISH TUMORS OR OTHER DEFORMITIES	When the incidence rates of fish tumors or other deformities exceed rates at unimpacted control sites or when survey data confirm the presence of neoplastic or preneoplastic liver tumors in bullheads or suckers.	When the incidence rates of fish tumors or other deformities do not exceed rates at unimpacted control sites and when survey data confirm the absence of neoplastic or preneoplastic liver tumors in bullheads or suckers.
BIRD OR ANIMAL DEFORMITIES OR REPRODUCTIVE PROBLEMS	When wildlife survey data confirm the presence of deformities (e.g., cross-bill syndrome) or other reproductive problems (e.g. eggshell thinning) in sentinel wildlife species.	When the incidence rates of deformities (e.g., cross-bill syndrome) or reproductive problems (e.g., eggshell thinning) in sentinel wildlife species do not exceed background levels in inland control populations.
DEGRADATION OF BENTHOS	When the benthic macroinvertebrate community structure significantly diverges from unimpacted control sites of comparable physical and chemical characteristics. In addition, this use will be considered impaired when toxicity (as defined by relevant, field-validated, bioassays with appropriate quality assurance/quality controls) of sediment-associated contaminants at a site is significantly higher than controls.	When the benthic macroinvertebrate community structure does not significantly diverge from unimpacted control sites of comparable physical and chemical characteristics. Further, in the absence of community structure data, this use will be considered restored when toxicity of sediment-associated contaminants is not significantly higher than controls.
RESTRICTIONS ON DREDGING ACTIVITIES	When contaminants in sediments exceed standards, criteria, or guidelines such that there are restrictions on dredging or disposal activities.	When contaminants in sediments do not exceed standards, criteria, or guidelines such that there are restrictions on dredging or disposal activities.

USE IMPAIRMENT	LISTING GUIDELINE	DELISTING GUIDELINE
EUTROPHICATION OR UNDESIRABLE ALGAE	When there are persistent water quality problems (e.g., dissolved oxygen depletion of bottom waters, nuisance algal blooms or accumulation, decreased water clarity, etc.) attributed to cultural eutrophication.	When there are no persistent water quality problems (e.g., dissolved oxygen depletion of bottom waters, nuisance algal blooms or accumulation, decreased water clarity, etc.) attributed to cultural eutrophication.
RESTRICTIONS ON DRINKING WATER CONSUMPTION OR TASTE AND ODOR PROBLEMS	When treated drinking water supplies are impacted to the extent that: 1) densities of disease-causing organisms or concentrations of hazardous or toxic chemicals or radioactive substances exceed human health standards, objectives or guidelines; 2) taste and odor problems are present; or 3) treatment needed to make raw water suitable for drinking is beyond the standard treatment used in comparable portions of the Great Lakes which are not degraded (i.e., settling, coagulation, disinfection).	For treated drinking water supplies: 1) when densities of disease-causing organisms or concentrations of hazardous or toxic chemicals or radioactive substances do not exceed human health objectives, standards or guidelines; 2) when taste and odor problems are absent; and 3) when treatment needed to make raw water suitable for drinking does not exceed the standard treatment used in comparable portions of the Great Lakes which are not degraded (i.e., settling, coagulation, disinfection).
BEACH CLOSINGS	When waters, which are commonly used for total-body contact or partial-body contact recreation, exceed standards, objectives, or guidelines for such use.	When waters, which are commonly used for total-body contact or partial-body contact recreation, do not exceed standards, objectives, or guidelines for such use.
DEGRADATION OF AESTHETICS	When any substance in water produces a persistent objectionable deposit, unnatural color or turbidity, or unnatural odor (e.g., oil slick, surface scum).	When the waters are devoid of any substance which produces a persistent objectionable deposit, unnatural color or turbidity, or unnatural odor (e.g., oil slick, surface scum).
ADDED COSTS TO AGRICULTURE OR INDUSTRY	When there are additional costs required to treat the water prior to use for agricultural purposes (i.e., including but not limited to, livestock watering, irrigation and crop-spraying) or industrial purposes (i.e., intended for commercial or industrial applications and noncontact food processing).	When there are no additional costs required to treat the water prior to use for agricultural purposes (i.e., including, but not limited to, livestock watering, irrigation and crop-spraying) and industrial purposes (i.e., intended for commercial or industrial applications and noncontact food processing).
DEGRADATION OF PHYTOPLANKTON AND ZOOPLANKTON POPULATIONS	When phytoplankton or zooplankton community structure significantly diverges from unimpacted control sites of comparable physical and chemical characteristics. In addition, this use will be considered impaired when relevant, field-validated, phytoplankton or zooplankton bioassays (e.g., *Ceriodaphnia*, algal fractionation bioassays) with appropriate quality assurance/quality controls confirm toxicity in ambient waters.	When phytoplankton and zooplankton community structure does not significantly diverge from unimpacted control sites of comparable physical and chemical characteristics. Further, in the absence of community structure data, this use will be considered restored when phytoplankton and zooplankton bioassays confirm no significant toxicity in ambient waters.
LOSS OF FISH AND WILDLIFE HABITAT	When fish and wildlife management goals have not been met as a result of loss of fish and wildlife habitat due to a perturbation in the physical, chemical, or biological integrity of the Boundary Waters, including wetlands.	When the amount and quality of physical, chemical, and biological habitat required to meet fish and wildlife management goals have been achieved and protected.

Source: IJC (1991).

1.2 GOAL OF OUR STUDY

The goal of this study is :

To describe and catalogue impaired uses in Lake Ontario and to identify the causal pollutants whose elimination or reduced loading will restore beneficial uses.

1.3 PRODUCTS OF THE STUDY

The products of the study are:

1. A set of guidelines for designation of uses as impaired and causal pollutants;
2. A small workshop to discuss preliminary findings of impairment of beneficial uses and their causal pollutants;
3. An inventory of known impaired uses, their locations and the pollutants responsible;
4. A summary document defining and identifying the most frequently and severely impaired uses, their locations, and the causal pollutants responsible.

1.4 APPROACH OF THE STUDY

The study consisted of 4 distinct stages:

Stage 1: Information gathering: interviews and literature search;

Stage 2: Development and application of critical appraisal guidelines;

Stage 3: Development and review of inventory; and

Stage 4: Formulation of findings and recommendations.

In Stage 1, information gathering, many of the key individuals involved in research, policy and programs on the Great Lakes, particularly on Lake Ontario, were interviewed. Three of us interviewed a total of about 100 experts during the period January to April, 1991. The aim of the interviews was to gather existing reports and data on Lake Ontario, both published and unpublished, and to identify additional sources of information. This personal approach helped to identify ongoing research and unpublished or soon-to-be published data; some 300 documents were identified. The results of the interviews are outlined in Chapter 3.

In addition to the interviews, on-line reference databases were searched using key words drawn from the list of impairments. This generated approximately 200 additional references. All 500 documents were then assembled, read and critically appraised using the guidelines discussed in Chapter 4.

Stage 2, the development and application of the critical appraisal guidelines, involved the evaluation of the 500 references against selected criteria; see Chapter 4.

Stage 3. The development and review of an inventory of impairment of beneficial uses, involved identifying where the impairment occurred, the contaminant(s) most likely responsible for the impairment and the source of the contaminant(s). The inventory was compiled by extracting relevant information from approximately 500 references. For example, references describing fish tumors are entered into the inventory under impairment of beneficial use #4 – Fish tumors and other deformities – as well as the reference number of

the article, the location of the impairment, the fish species, the description of the impairment, the source of contaminant, the pollutant(s) responsible and the type of reference. The complete inventory is described in Chapter 5. The inventory was designed to be simple and concise in order to summarise large quantities of information, and to allow identification of the causal pollutants. For some of the impairments, such as #5 – bird or animal deformities and reproductive problems – over 50 references were found; for #2 – fish tainting – no documents were identified.

Three selected impairments, were reviewed at a workshop of 30 experts on Lake Ontario research and policy: #4, Fish tumors and other deformities; #5, Bird or animal deformities or reproductive problems; and #10, Beach Closures. This workshop was designed to test the completeness and accuracy of the information listed in the three impairments, and to provide feedback that could be applied in developing the remaining impairments of beneficial uses.

Stage 4 reviews the inventory to identify the causal pollutants, and to produce an overall summary of the findings of the study. This stage developed the links between impairments and pollutants to generate a final list of causal pollutants. The criteria used to identify the causal pollutants involved a combination of presence at elevated levels in Lake Ontario and association with an impairment by the weight-of-evidence and of recognised causal links.

Summaries were produced documenting the most frequently occurring impairments of beneficial use, the beneficial use most severely impaired, and the location in Lake Ontario

with the greatest number of impairments.

1.5 BOUNDARIES OF THE STUDY

Several decisions were made regarding the boundaries of the study. Geographically, the study covers both nearshore and open water areas of Lake Ontario. Major tributaries such as the Niagara River are also included as inputs. The study treats the St.Lawrence as the outlet to Lake Ontario (Lake Ontario Secretariat, 1990).

Temporally, the study focused on data from 1970 - 1991. Information on bird, animal and fish populations and deformities is often compared with earlier historical records. These historical data were also included in the inventory for comparitive purposes when it seemed appropriate.

The impairment of beneficial use #14 — the loss of fish and wildlife habitat, is not considered in this study; a survey on this impairment has been planned by the United States Environmental Protection Agency, to be conducted by the U.S. Fish and Wildlife Service.

1.6 STUDY TEAM

A new Institute of Environment and Health has recently been created by McMaster University and the University of Toronto. At Toronto, it is being developed at least initially within the mandate of the Institute for Environmental Studies, while at McMaster it has been recognised as a separate Institute. The Institute of Environment and Health has strong involvement from individuals from toxicological, medical, engineering, ecological and sociological disciplines. Interdisciplinary research, teaching and community service

emphasises both the reactive correction of pollution and proactive prevention.

The principal investigators for the study were Henry Regier, Director of the Institute for Environmental Studies at the University of Toronto and Evert Nieboer, Professor of Toxicology, McMaster University. The Project Manager was Sarah Rang, Environmental Economics International; the Senior Researchers were John Holmes, University of Toronto and Sandy Slota, McMaster University; and the Junior Researcher was Nathalie LaViolette, University of Toronto. Dr. Douglas Bryant, Assistant Professor in Biochemistry McMaster University, participated as a co-investigator.

1.7 OVERVIEW OF THE REPORT

The report is organised into 8 Chapters. Chapter 1, Introduction, provides a description of the study framework and approach; Chapter 2, Lake Ontario, provides background information on the hydrology, land use and ecology of Lake Ontario; Chapter 3, Public Consultations, outlines the results of interviews and findings from the workshop; Chapter 4, Critical Appraisal Guidelines outlines the history, need and development of critical appraisal guidelines; Chapter 5, Inventory of Impairments of Beneficial Uses, summarizes the information into the impairment categories and identifies where the impairments occur, the contaminant responsible for their occurrence and the source of the contaminant; Chapter 6, Discussion, provides a summary of the inventory and a discussion of the major findings of the study; Chapter 7, Recommendations; and Chapter 8, Literature Cited, provides the cited references. A series of Appendices provides futher details.

CHAPTER 2 LAKE ONTARIO

2.1 PHYSICAL FEATURES

Lake Ontario lies at the downstream end of the chain of the Great Lakes. At the upstream northwest end of the chain the waters drain the hard and infertile Laurentian Shield. Southeastward towards Lake Ontario much of the basin consists of softer, more fertile sedimentary lands. At the far eastern edge, the waters that leave this chain of lakes pass over a sill of the Laurentian Shield. This sill has been resistant to erosion, even by continental glaciers and has prevented ingress of saline water into Lake Ontario.

Physiographically, Lake Ontario has two major basins. The main south western basin extends from the west end to Mexico Bay in the northeast (Figure 2.1) and contains the deepest waters. The smaller northeastern Kingston Basin lies downstream from the Duck-Galloo Island sill and is generally less than 30 m deep, except where the St. Lawrence trough cuts through it (Sly, 1991). Three minor sedimentation basins are recognized in the main basin, from west to east; these are the Niagara, Mississauga, and Rochester basins. Compared to the other Great Lakes, the bottom of Lake Ontario is relatively smooth and featureless, except toward the northeast where a large peninsula and numerous islands are complemented by some underwater morainic deposits. The south shore slopes more steeply than the north shore (Stevens, 1988). Shoreline development (i.e., the length of shoreline relative to the circumference of a circle of area equal to that of the lake) and hence aquatic habitat diversity, is greatest in the eastern part (Sly, 1991). Only 8% of the lake area is less than 10 m deep and 16% is less than 20 m deep (Sly, 1991). Most of these ecologically more productive shallow areas are located in the Kingston Basin and a thin strip around the perimeter of the main basin.

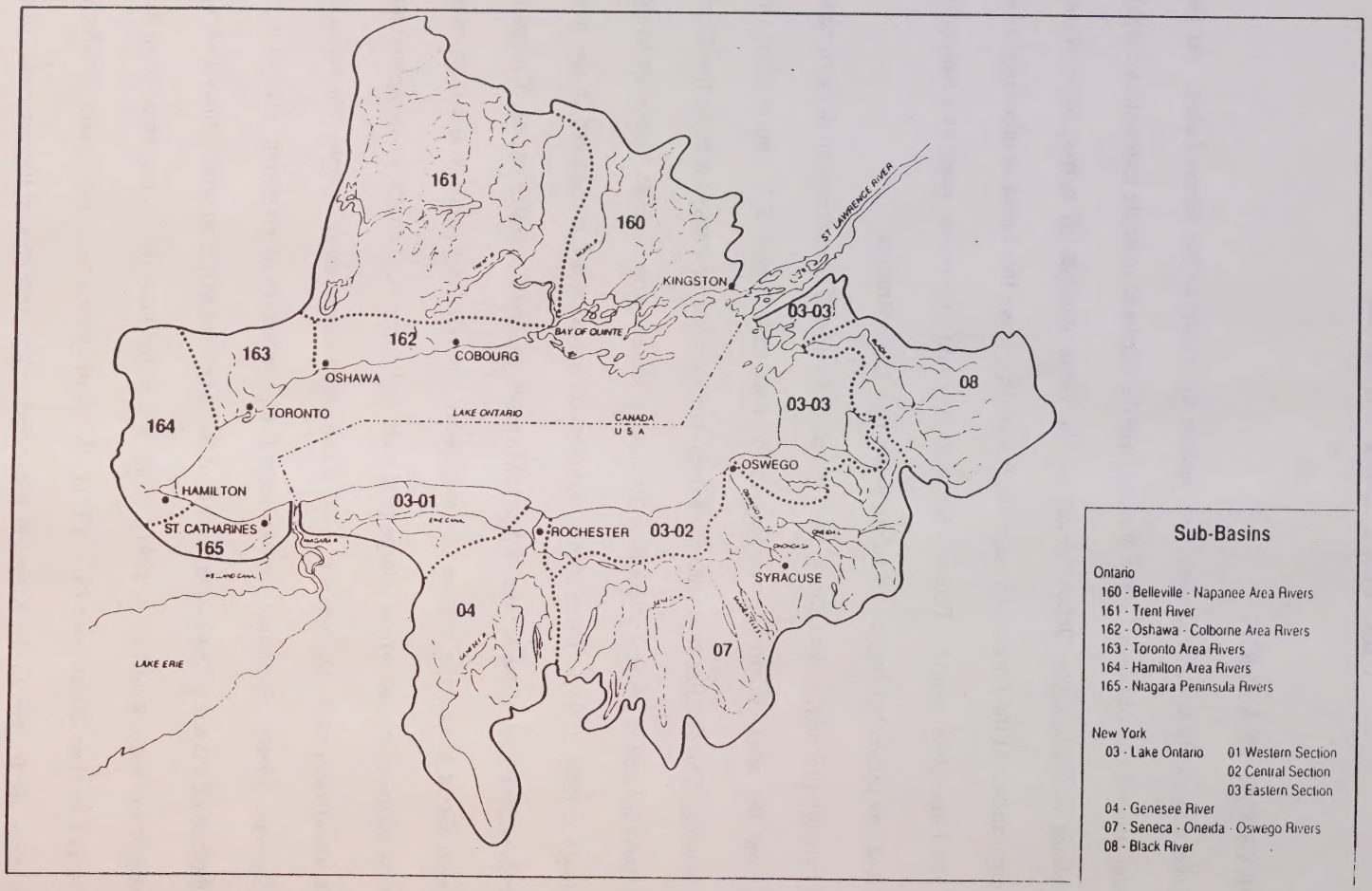

Figure 2.1. Lake Ontario and its surrounding drainage basin; from Lake Ontario Secretariat (1990).

In terms of surface area, Lake Ontario (Figure 2.1) is the smallest of the Great Lakes. At 18,960 km^2 it represents 7.8% of the total Great Lakes surface area, with a land drainage area of 64,030 km^2, which represents 12.2% of the total drainage area in the Great Lakes basin (Botts and Krushelnicki, 1987; Sly, 1991). Lake Ontario is the second deepest lake with an average depth of 86 m and a maximum depth of 244 m, but its volume, 1,640 km^3, surpasses only Lake Erie (Aubert and Richards, 1981).

2.2 HYDROLOGY

Outflow from Lake Erie dominates the hydrology of Lake Ontario. The average inflow from Lake Erie (1900 to 1983) through the Welland Canal and the Niagara River is 5,800 m^3/s (Stevens, 1988). This is equivalent to about 85% of the total average outflow of 6,800 m^3/s down the St. Lawrence River (Stevens, 1988). An additional 9% is contributed by the Oswego, Trent, Black, and Genesee Rivers with average flows, between 1974 and 1982, of 230, 160, 123, and 97 m^3/s respectively (Stevens, 1988). Along the north shore Bronte Creek, and the Credit, Humber, Don, and Ganaraska Rivers have a combined flow of about 22 m^3/s into the main basin (Sly, 1991), with the remaining tributaries and point sources accounting for a further 180 m^3/s. Direct precipitation to the lake surface provides approximately 500 m^3/s while evaporation represents an average loss of 530 m^3/s (Stevens, 1988).

The theoretical "residence" time of water in Lake Ontario is 7.6 years based on the figures reported for volume, inflow, and outflow. Practically, however, residence time in the hypolimnion may be as long as 20 years because there is relatively little mixing during the

summer months when the water column is stratified (Stevens, 1988). In contrast, residence time in the epilimnion may be as little as 3.5 years, if all inflow from the Niagara River were assumed to remain in the epilimnion (Stevens, 1988). Of course, these concepts are abstractions, but for some idea of the time scales necessary to detect changes in water quality these figures can be compared with Lake Superior where the residence time is 191 years (Colborn et al., 1990).

Water surface level in Lake Ontario is also a function of isostatic rebound and of deliberate regulation of the water level by means of a dam on the St. Lawrence River. The eastern end of the lake, the area with the Laurentian Shield sill, is rising approximately 0.35 m per century relative to the west end (Sly, 1991). The Lake Ontario outflow is regulated by the structures the St. Lawrence Seaway and associated hydroelectric facilities. Regulation in recent decades has resulted in smaller water level fluctuations compared to those of the other Great Lakes.

2.3 CIRCULATION AND WATER CURRENTS

The linkage between thermal structure of the water column and currents is an important feature of the dynamics of water movement in Lake Ontario. In the absence of strong winds in spring (the usual case), water in nearshore areas warms faster than in offshore areas; as a result of the temperature differential a vertical thermocline forms to separate isothermal warm nearshore water from isothermal colder offshore water. This structure prevents mixing of near- and offshore waters while it persists and is known as a thermal bar. The thermal bar moves offshore as spring heating continues (Boyce et al., 1989); the

maximum extent of the thermal bar is 20-25 km off the north shore and about 10 km off the south shore (Stevens, 1988). Usually it dissipates about mid-June, signalling the beginning of lake-wide vertical stratification (Rodgers, 1987).

The thermal bar phenomenon affects the fate and transport of contaminants from nearshore inputs. Except for atmospheric deposition, any spills from ships and the outflow from the northerly top of the Welland Canal, the inputs into Lake Ontario are from shore and stream outflows. During periods of thermal bar persistence, contaminants and pollutants from nearshore inputs may be trapped in a narrow band around the perimeter of the lake with relatively little onshore/offshore mixing. Water quality can be noticeably degraded nearshore relative to offshore waters (Boyce et al., 1989). This "ring of pollution" has implications for aquatic organisms. For example, salmonines such as rainbow trout move alongshore within the thermal bar in spring and tend to disperse offshore as temperatures in it exceed 10 °C (Haynes et al., 1986). Thus fish may be exposed to harmful concentrations of contaminants during a critical reproductive phase in their life cycle.

Thermal bar formation creates a pressure gradient which tends to push warm water offshore (Boyce et al., 1989). This movement is deflected by the Coriolis force resulting in a counter-clockwise flow called a "coastal jet". Shore-parallel currents persist throughout the stratified period. These currents are sensitive to the direction and magnitude of prevailing winds. Water moves predominantly eastward along the south shore and westward along the north shore (Stevens, 1988; Sly, 1991) with greater transport occurring along the south shore because of the outflow into the St. Lawrence River. Overall, there is a persistent gyral circulation in Lake Ontario with transport 10 times greater than the Niagara River flow

(Boyce et al., 1989). Thus material dissolved in Niagara River waters is circulated to all parts of the lake, though a fraction of the Niagara water may flow rather directly from the Niagara River mouth eastward along the south shore and exit into the St. Lawrence River.

The frequency of up- and down-welling episodes is a second important feature of water dynamics in Lake Ontario. Winds commonly blow along the long axis of the lake from southwest to northeast or a long a shorter axis from northwest to southeast. These winds cause transient up-welling along the north shore and down-welling along the south shore in summer because winds from the northwest pre-dominate, while the opposite occurs in winter (Stevens, 1988). Up-wellings are particularly significant if they occur during the summer stratified period because they usually bring up hypolimnetic waters that are much colder than prevailing nearshore conditions. Pelagic coldwater fauna are carried to inshore areas during such episodes, thus disrupting the isolation of the hypolimnion.

The effects of the coastal jets and shore-parallel currents are generally restricted to within 10 km of the shoreline. These currents are therefore important when assessing the fate and transport of contaminants from nearshore inputs (Stevens, 1988). Contaminants may be carried tens of km along the shore with little onshore/offshore mixing. The distributions of mercury, PCB, and mirex in the surficial bottom sediments of Lake Ontario clearly demonstrate the significance of the Niagara River as a source and the effect of the usual eastward boundary current along the south shore, but also the episodic westward displacement of the Niagara River plume as a result of more occasional wind-generated phenomena (Thomas, 1983).

2.4 POPULATION AND LAND USE

Approximately 8 million people live in the Lake Ontario drainage basin. In contrast to the other Great Lakes, most of the population in this basin, 64%, live on the Canadian side (Sly, 1991). The greatest concentrations of industry and humans, and hence the highest intensity of activities likely to impair beneficial uses, occur in Hamilton, upstream of the mouth of the Niagara River, in Rochester and in Toronto.

Land use in the drainage basin of Lake Ontario reflects the differences in population distribution (see Section 5.12). Agriculture accounts for 49% of the area in Ontario compared to 33% in New York (Botts and Krushelnicki, 1987). In contrast, forest covers 53% of the relevant area of New York and 42% of Ontario. Major differences in shoreline use are found in commercial and residential categories. Of the shoreline in Ontario 18% is commercial and 25% residential, whereas in New York the comparable figures are 8 and 40% (Botts and Krushelnicki, 1987). With nearly twice the population in Ontario compared to New York, these data indicate much higher densities in urbanized areas of Ontario.

European settlement in the Lake Ontario basin began in the 1700s (Sly, 1991). Since this time the basin lands and the shoreline have become heavily agriculturalized with the attendant problems of environmental degradation, species extirpations, and introductions of exotics (see Sections 5.3 and 5.13). Humans use the waters of the lake for drinking, industrial supply, vessel transport, swimming, fishing, and other recreational activities (see Sections 5.1, 5.9, 5.10, and 5.12). These waters receive a variety of municipal and industrial wastes produced within the basin and transported into the lake from outside by hydrologic and atmospheric currents. Inputs from 15 municipal sewage treatment plants (12 in Ontario

and three in New York) and two industrial facilities (one each in Ontario and New York) are considered significant sources of persistent toxic substances in Lake Ontario (Lake Ontario Secretariat, 1990). The Niagara River is a significant source of toxics from within its drainage basin and from the entire Great Lakes drainage basin upstream (see Sections 5.1, 5.5, 5.6, and 5.8).

Environmental degradation reached its nadir in the 1960s. Since this time conditions have improved considerably as a result of government regulation of uses, changes in industrial and agricultural practices, and some de-industrialization as economies change. But conditions are still poor in many localized nearshore areas; and in many cases the degradation is due to past as well as current practices. Continuing inputs of contaminants from sediments, landfills, industry, municipalities, and the atmosphere are having harmful effects on fisheries (Cairns et al., 1984), wildlife populations (Colborn et al., 1990), benthos, phytoplankton, and zooplankton populations (Government of Canada, 1991a,b).

CHAPTER 3 PUBLIC CONSULTATIONS

3.1 INTRODUCTION

Two of the four products of the present project are: (1) an inventory for Lake Ontario of known impaired uses, their locations and the pollutants responsible; and (2) a small workshop to discuss preliminary findings of impairment of beneficial uses and their causal pollutants. It was clear at the outset of our study that we would need help from Great Lakes' scientists and experts to identify the many data sets and scientific studies pertinent to our inventory of toxic chemicals and associated effects in Lake Ontario. Extensive consultation was achieved through two different methods: personal interviews to determine relevant databases and studies and a workshop to assess our interpretations of the material. A synopsis of these communication and consultation efforts is provided in this chapter.

3.2 INTERVIEWS

3.2.1 Experts Interviewed

The summary provided in Table 3.1 illustrates that over 100 individuals were interviewed. The largest group consulted were experts employed by government departments and agencies. The others were distributed among non-governmental organizations; universities; industry; private consultants; and persons involved with RAP programmes. A complete list of experts contacted is presented in Appendix A.

Table 3.1. Affiliation of experts interviewed.

Affiliation	Number of Individuals Interviewed	Affiliation Specifics
Non-Governmental Organizations	15	Board of Trade; Canadian Environmental Law Association; Canadian Manufacturers Association; Great Lakes United; Greenpeace; IJC; Pollution Probe; Rawson Academy; World Wildlife Fund.
Government Agencies	50	Environment Canada, Department of Fisheries and Oceans, Health and Welfare Canada, Agriculture Canada, Public Works Canada, Ontario Ministries of the Environment, Health, Agriculture and Food, Natural Resources; U.S. Fish and Wildlife Service; N.Y. Department of Health, N.Y. Department of Environmental Conservation; Monroe County Departments/ Authorities; Rochester Region; Grand River Conservation Authority; Cities of St. Catharines and Toronto.
Universities	20	Brock, Guelph, McMaster, Queen's, SUNY (Buffalo), Trent, Toronto, Waterloo, Western Ontario.
Other	>15	Consultants, Finnie Distributing, Kodak, Molson Breweries, Ontario Hydro, Roswell Park (NY); RAP programmes at Bay of Quinte, Hamilton, Niagara, Port Hope and Toronto.

3.2.2 Information Gained and Comments by Experts Interviewed

The personal interviews were helpful in identifying important contacts, published and unpublished studies and ongoing research. For those impairments of beneficial uses for which written documentation was scarce, the referral to persons with pertinent information or data was crucial. Clearly, the Lake Ontario database extends much beyond the formal literature. We sensed that there is much enthusiasm for lakewide management plans. Persons interviewed also shared their perspectives and concerns. A summary of their views and opinions as expressed on a number of topics, is outlined below; the order of presentation reflects neither relative importance nor priority.

(i) *Concept of Causal Pollutants.* Many felt that the establishment of direct causal links to impairments was a very difficult task. It was mentioned that this approach to remedial action had been abandoned by a number of RAP committees. A number of individuals suggested that grouping pollutants into classes might help since impairments have multiple causes. The chemical-by-chemical approach was judged to be slow and ineffective. It also ignores possible synergistic effects. Each chemical might be considered guilty until proven innocent, especially in relation to preventive strategies.

(ii) *Critical Appraisal of Evidence.* It was felt that researchers/authors needed to address the validity and reliability of their data more extensively (consideration of sample size, bias, observed variability, manner in which outcome is assessed, use of controls).

(iii) *Intersectoral/Interdisciplinary Cooperation.* The need for various experts to work together (e.g. wildlife scientists and toxicologists) was identified. Although an interdisciplinary approach was felt essential, the logistical and conceptional difficulties were acknowledged. In addition, four-party cooperation in research was viewed as necessary but difficult due to suspicion, different perspectives and slowness of building consensus. The participation of municipalities in pollution issues and activities was deemed crucial.

(iv) *Historical Trends.* The need for establishing temporal trends of contaminant concentrations (in water, sediments, organisms, etc.) and associated effects was emphasized. Even though some data are available, such as for PCBs and nutrient levels in Lake Ontario compartments, collection of more data is warranted. Historical data on beach closings and floods were also viewed as important in the interpretation of changes, in and management of Lake Ontario.

(v) *Issues Beyond Toxics.* The opinion was offered that landuse, wildlife management criteria, and habitat concerns are not receiving enough attention.

(vi) *Tainting of Fish and Wildlife and Degradation of Aesthetics.* A number of individuals stressed that these impairments are difficult to assess because of subjectivity and differences in perception.

(vii) *Sustainable Agriculture.* There is a need to reverse the processes of physical destruction of soil and losses to Lake Ontario. Most added costs to agriculture appear to be in the form of remedial or control activities related to erosion.

(viii) *Degradation of Benthos.* The wording of this impairment leaves unclear the basis for this value judgment.

(ix) *Congener Specificity.* Congener specific concentrations were advocated to allow the establishment of toxicologic equivalents (e.g. of PCBs, dioxins).

(x) *Lack of Data.* The scarcity of data for a number of areas was identified: wildlife toxicity; clear cause-and-effect linkages; psycho-social impact of Lake Ontario contaminants; the impairments of degradation of aesthetics; tainting of fish and wildlife; added costs to agriculture or industry; and loss of fish and wildlife habitat.

(xi) *Educational Needs.* If environmental issues are to become mainstream to societal priorities, sensitization of individuals through education, especially of young people, must receive increasingly high priority.

3.3 WORKSHOP

3.3.1 Participants and Agenda

Appendix B provides a list of the 32 individuals that participated in the day-long workshop held at McMaster University, June 11, 1991. The participants were scientists and other experts from government departments and agencies, the IJC, universities, non-governmental organizations and industry.

The agenda for the workshop is reproduced as Appendix C. Prior to the workshop, all participants received copies of the following documents: (1) An introductory chapter describing the background, goals, products and approach to the study of *Impairment of*

Beneficial Uses in Lake Ontario; (2) A description of the approach to a review and critical appraisal of the available evidence and the establishment of a computer-based inventory; and (3) Summaries of the inventory for three Impairments, i.e., #4 – Fish Tumors or Other Deformities, #5 – Bird or Animal Deformities or Reproductive Problems, and #10 – Beach Closings. Highlights of this written material were presented orally by the authors, followed by questions of clarification and discussion. Mike Gilbertson of the IJC provided oral summaries of the feedback and comments of the participants. The workshop concluded with an extensive discussion dealing with both specific items and general principles and perspectives.

3.3.2 Discussion Summary

In order to facilitate a factual account of the various discussions that took place at the workshop, the proceedings were recorded with the permission of the participants. Tapes of the recordings are available to interested parties. The views and opinions expressed by the participants are outlined below; see Chapter 6 for a discussion of these views.

(i) *Availability and Updating of the Lake Ontario Inventory*

Considerable interest was expressed in making the Inventory available to others and to keeping it current. Clearly, the latter depends on long-term financial support, albeit at low cost. The suggestion was made to explore the merging of the Inventory with two existing databases maintained by Tim Kubiak (U.S. Fish and Wildlife Service, Lansing, MI) and Theo Colborn (World Wildlife Fund, Washington, D.C.). The IJC might be interested in such an

ongoing literature database, as might the Great Lakes Health Effects Program of Health and Welfare Canada. Governments appear to be defining new roles as facilitators and might welcome a non-governmental database with appropriate management. The database could also serve as a Central Registry for Great Lakes Research. Both aspects would facilitate Lakewide Management, especially if contaminant sources were identified as well.

(ii) *Critical Appraisal (Rules of Evidence)*

Near consensus was evident about the need for guidelines that facilitate sound judgement and argument about the reliability of our knowledge and its use in cause-and-effect arguments. For example, what histopathologic criteria should be used to standardize fish tumor diagnosis? Less effort is spent today on identifying potential cause-and-effect linkages (i.e., causal hypotheses; answers to what is going on). Field observations such as bill deformities and tumors are as important as carefully collected analytical data. Rules of evidence are needed to help us construct a believable story, employing not only analytical data, but also toxicologic, ecologic and, if available, human epidemiologic evidence. This comprehensive approach was referred to by several participants as the 'New Science'. Its link to the regulatory process and traditional methods is the cause of tension; we need to learn how to facilitate a marriage.

(iii) *Rating Scale for Publications Reviewed for Inclusion in the Inventory*

A considerable proportion of the participants felt that some kind of global numerical rating or aggregate assessment should be assigned to each publication reviewed in the Inventory. This, it was thought, followed directly from the critical appraisal exercise. In response, the study team explained why they opposed this. The objective of the study was

not to credit or discredit specific papers, but rather to examine the connection and relationships among the papers to derive the list of causal pollutants. The information entered on the data entry form, or the lack of it, may help the user of the Inventory to assess the quality of evidence with respect to the present issue. Such individual assessment about reliability would be somewhat facilitated by comments in the data-entry form memo concerning specific strengths or weaknesses exhibited by the document in question.

(iv) *Causal Pollutants*

Substantial discussion took place on how to designate causal pollutants. A synthesis of the many arguments presented may be summarized as a weight-of-evidence approach. A decision to designate a candidate pollutant as causal might incorporate evidence such as: field observations (epidemiologic, analytical, toxicologic and ecologic); evidence supporting associations or cause-and-effect linkages; the toxicologic profile of the substance (preferably as a member of a family of pollutants); and actual input/loading data from all sources. Loading data are currently being obtained under the Lake Ontario Toxics Management Plan. Such an approach would require that decision makers look beyond Lake Ontario to other waters and to both field and laboratory studies. The overall decision would consider a combination of a number of lines of evidence.

There was some support for the view that compounds lacking toxicologic information should not be discharged into lakes at all. Demonstration of a causal link is not required. The onus should be on the polluter to demonstrate that a substance is harmless and can be safely discharged.

(v) *The Decision-Making Process*

In the capacity as an ordinary citizen, the scientist can facilitate the acceptance of combined scientific evidence in decision-making and shift the regulators away from over-emphasizing the requirement of hard proof of causality. He or she has a mediatory role to play, and thus alleviate tension. The emotionally-charged stance regarding pollution of the Great Lakes, and the demand for unreasonable proof by conservative and reactionary groups as a prerequisite to any clean up/preventive action, must be addressed by balanced discussion.

(vi) *Off-Shore Versus On-Shore Impairments*

Lively debate took place on the relationship between near-shore and open-water effects and controls. The views expressed included the following comments: focus only on off-shore data and effects; expect that offshore impairments may integrate inshore Areas of Concern, AOC, impairments and associated contaminant concentrations; use contaminant-fate modelling to predict open-water values; fish and wildlife are not stationary and tend to migrate/integrate into the entire area of a lake; an ecosystem approach is mandated; and RAPs are parts of a lake-wide management plan that are administered locally. According to these comments the geographic boundaries of LAMPs in some way include the RAP Areas of Concern.

vii) *Advice on Impairments #4 (Fish Tumors or Other Deformities), #5 (Bird or Animal Deformities or Reproductive Problems) and #10 (Beach Closings)*

Advice focused on additional sources of information and data. In terms of Impairment #4, considerable discussion took place about the quality control measures required in the diagnosis of fish tumors. Standardization of histopathologic criteria employed and inter-laboratory comparison were encouraged to reduce variability and promote harmonization for

interpretation of results. Only through cooperative efforts can tumor counts and inter-study comparisons gain relevance.

The discussion of Impairment #5 addressed the use of contaminant concentration and biochemical marker data. It was felt that these serve at least as indicators of exposure. In addition, some of the induced-enzyme phenomenon are quite substance-specific. In a weight-of-evidence approach, the combination of such data with observed health effects strengthens associations with causal pollutants. Such an approach was encouraged.

For both Impairments #4 and #5, the need for adequate controls was deemed crucial in order to demonstrate that the incidence of tumors or deformities are indeed in excess. This discussion again reinforced the need for rules of evidence in assessing the reliability of published studies.

In terms of beach closings, the definition of "beach" was reviewed. "Beach" implies a place where human activity involves partial or whole-body water contact and could be a dock or rocky outcrop. This interpretation may have bearing on the posting of swimming restrictions. Many participants wondered whether beaches had been closed for reasons other than bacterial contamination. For instance, does chemical contamination cause impairment of beach use?

For all three impairments, continuous data collection over long periods of time was deemed essential. Long-term trends are extremely important for Impairments #4 and #5. Unfortunately, much of the data available is quite dated and may not reflect today's aquatic environment and contaminant impact.

CHAPTER 4 APPROACH TO A REVIEW AND CRITICAL APPRAISAL OF THE AVAILABLE EVIDENCE AND THE ESTABLISHMENT OF A COMPUTER-BASED INVENTORY

4.1 INTRODUCTION

In this chapter, details are provided about the approach taken to review the pertinent literature and construct the computer-based inventory of publications describing impairment of beneficial uses of Lake Ontario and associated causal pollutants. To ascertain the reliability of the evidence reviewed, critical appraisal guidelines are helpful. These are not readily available, nor self-evident for the diverse subject areas covered by our report. Our first task, therefore, was to formulate rules of evidence suitable for a critical assessment of studies describing relevant experiments or observations falling within the domains of analytical measurements, ecology, epidemiology of pollution-related health effects and toxicology.

4.2 EPIDEMIOLOGICAL CRITERIA FOR INFERRING CAUSALITY

The determination of quality of evidence goes beyond an application of simple statistical tests, especially when cause-effect linkages and weight-of-evidence issues are considered such as in the present report. As pointed out by Fox (1989) and Gilbertson (1989a), criteria inferring causality go back to the 1860s and are attributed to the bacteriologist Robert Koch who related a number of diseases to specific microbes. Particularly relevant to our purpose is the formulation of criteria by an Advisory Committee of the U.S. Surgeon General in its deliberations about smoking and disease, with further applications by others (Department of Health, Education, and Welfare, DHEW, 1964; Hill, 1965; Fox, 1989; Gilbertson, 1989a). The five criteria identified are reproduced in Table 4.1.

Hill (1965), an epidemiologist, modified and expanded these criteria to include nine "viewpoints" that are useful in upgrading an *association* to a *cause* of a disease (Table 4.2). The list in Table 4.2 differs from that in Table 4.1 by the addition of dose-response relationship, plausibility, direct experiment and analogy to provide corroborative evidence for causation. A quotation from Hill (1965) clearly explains the manner in which the guidelines in Table 4.2 should be employed and that dogmatism must be avoided.

> "Here then are nine different viewpoints from all of which we should study association before we cry causation. What I do not believe - and this has been suggested - is that we can usefully lay down some hard-and-fast rules of evidence that *must* be obeyed before we accept cause and effect. None of my nine viewpoints can bring indisputable evidence for or against the cause-and-effect hypothesis and none can be required as a *sine qua non*. What they can do, with greater or less strength, is to help us to make up our minds on the fundamental question - is there any other way of explaining the set of facts before us, is there any other answer equally, or more, likely than cause and effect?"

Hill (1965) continues to explain the relationship of his viewpoints or guidelines to tests of significance.

> "No formal tests of significance can answer those questions. Such tests can, and should, remind us of the effects that the play of chance can create, and they will instruct us in the likely magnitude of those effects. Beyond that they contribute nothing to the 'proof' of our hypothesis."

To complete this historical perspective of critical appraisal, it is relevant to indicate how Sackett (1980) has modified and ranked Hill's viewpoints in a slightly different way (Advisory Council on Occupational Health and Safety, ACOHS, 1983). The nine diagnostic tests for causation are arranged in Table 4.3 in decreasing order of importance.

Table 4.1. Criteria for inferring causality[a].

i	**Time Order:** does the exposure precede the manifestation of the disease?
ii	**Strength of association:** is there a statistically significant increase in the incidence of the disease in exposed populations compared to unexposed populations?
iii	**Specificity:** are the signs and symptoms only manifested with exposure to the suspected agent and is the agent linked only to specific signs and symptoms?
iv	**Consistency:** do similarly conducted studies in different times and places yield similar relationships?
v	**Coherence:** does the supposed relationship make biological sense and conform to existing knowledge?

[a]Adapted from Gilbertson (1989a), with DHEW (1964) as the original source.

Table 4.2. The Hill "Viewpoints" Diagnostic of Causation[a], in decreasing order of importance.

i	**Strength**, magnitude of effect.
ii	**Consistency**, has it been observed by others elsewhere?
iii	**Specificity**, of sites, groups, work, etc.
iv	**Temporality**, timing and duration of exposure; consideration of latency.
v	**Biological Gradient**, dose-response relationship,
vi	**Plausibility**, does it make biological sense?
vii	**Coherence**, agreement with other knowledge about the disease,
viii	**Experimental Confirmation**, e.g. do preventive actions work?,
ix	**Analogy**, judge by comparison to similar evidence,

[a]From Hill (1965).

Table 4.3. A framework for determining whether an association revealed in a study is causal in nature[a].

i	Does There Exist Evidence on the Subject from True Experiments in Humans, i.e. from Randomized Control Trials?
ii	Is the Association Strong?
iii	Is the Association Consistent from Study to Study?
iv	Is the Temporal Relationship Correct?
v	Is There a Dose-Response Gradient?
vi	Does the Association Make Epidemiologic Sense?
vii	Does The Association Make Biologic Sense?
viii	Is the Association Specific?
ix	Is the Association Analogous to a Previously Proven Causal Relationship?

[a]From ACOHS (1983), with Sackett (1980) as the original source.

In the context of reviewing old and new epidemiologic knowledge, Sackett and his colleagues recommend the modified Hill criteria in Table 4.3 as a suitable framework for determining whether an association revealed in an epidemiologic study is causal in nature (ACOHS, 1983). They also clarify that: "Two variables (e.g. exposure to agent (X) and the incidence of disease (Y)) are said to be associated when they are related to one another in some fashion. A causal relationship arises, on the other hand, when the disease is shown to be induced by the environmental agent".

In the next section, we address the formulation and application of comparable criteria for quality of evidence for three other subject areas pertinent to this report, namely analytical measurements, ecology and toxicology. Some of the required criteria are related to those considered for the epidemiologic context.

4.3 QUALITY OF EVIDENCE ASSESSMENT

Critical appraisal is a scientific skill that needs to be developed and practiced. As already intimated, in our study four domains serve as sources of the material covered by our review, namely analytical measurements, ecology, epidemiology and toxicology. Evidence falling in these four domains needs to be reliable and valid if it is to serve in cause-effect linkage arguments or weight-of-evidence decisions. To achieve this goal, simple questions are formulated for each subject area to assist reviewers in assessing studies. It should be emphasized that the development of critical appraisal guidelines constitutes an ongoing, iterative process.

4.3.1 Critical Appraisal Questions for Epidemiologic Studies

The task at hand is to transform the pertinent diagnostic features of causation discussed in Section 4.2 to questions that permit an assessment of the quality of the evidence reported in a publication. This can be achieved by first examining the design and execution of an epidemiologic study, since the inherent quality of evidence depends on this (ACOHS, 1983; Taylor et al., 1988). Additional queries pertain to descriptors about the population, exposure and the statistical treatment. What population was studied and what was its size? How long were the exposure and the follow-up study? Was exposure assessment of the study population quantitative or qualitative; what was used was an exposure index? Was the effect on the population statistically significant (Error of the First Kind, with α the probability of rejecting the null hypothesis when it is true) and the power sufficient (equal to 1-ß, with ß the probability of an Error of the Second Kind which corresponds to accepting the null

hypothesis when the alternative is true, or detecting a real effect)? Questions concerning consistency, coherence, experimental confirmation and analogy are particularly suitable when a number of studies are compared and an overview is formulated. A complete listing of the actual questions used in our Inventory is provided in Section 4.5.

4.3.2 Critical Appraisal Questions for Analytical Measurements

In our review of papers reporting analytical data, annotations were made concerning compliance with the quality assurance measures outlined below. The specific questions posed for this purpose are given in Section 4.5.

Traditionally, estimates of accuracy and precision combined with statistical analyses have served to validate analytical results. In the last decade or so, the problems of inadvertent contamination of specimens during collection, handling and storage, and of samples during the actual analytical procedures, have awakened analysts to the need to implement strict quality assurance measures (Nieboer and Jusys, 1983). Consequently, publications need to provide information about specimen collection, handling and storage, about sampling (is it representative?) and the analytical method employed. Additional questions include:

(a) were reference materials and unpolluted (background) samples analyzed;

(b) was a performance comparison made with a reference or independent method;

(c) did the authors provide information about detection limit and precision; and

(d) was the apparent accumulation of the pollutant or its metabolite in the organism, target organ or matrix examined and analyzed relative to the control or reference situation statistically significant?

4.3.3 Appraisal Questions for Toxicologic Critical Studies

The need for quality assurance is especially critical in toxicology because of the possibility of biologic and experimental variation. Implementation of toxicologic critical appraisal guidelines is built into the preparation of Material Safety Data Sheets (MSDSs) as promulgated by the legislation on the Workplace Hazardous Materials Information System, WHMIS. As part of Canada's Hazardous Products Act, the *Controlled Products Regulation* (CPR) carefully specifies the evidence that is acceptable for MSDSs in terms of teratogenicity, embryotoxicity, reproductive toxicity, mutagenicity, carcinogenicity and sensitization of skin and the respiratory tract, among other effects. For nearly all experimental categories, the CPR adopts the Organization for Economic Cooperation and Development, (OECD) Guidelines for Testing of Chemicals (OECD, 1981). OECD specifies acceptable conditions and protocols pertaining to the experimental animals, number, sex, housing etc.; the test conditions, e.g., dose, observation period; clinical examination; pathology; and data treatment and reporting. To illustrate the type of questions that need to be posed or experimental information sought concerning cause-effect linkages, we focus specifically on teratogenic effects (Barlow and Sullivan, 1983; Wang and Schwetz, 1987). Consideration of the following experimental parameters assist in the evaluation of teratogenic potential:

(a) the number of animal species affected;
(b) the route of administration;
(c) the ratio of embryotoxic dose/maternal toxic dose;
(d) multiplicity and nature of developmental effects; and
(e) distribution of effects within and between litters.

In the species studied, are the malformations rare (specific) or common (non-specific)? Is the teratogenic mechanism understood?

The specific questions directed at toxicologic studies in the context of the present review reflect the pertinent elements of the quality assurance issues described or inferred in the previous paragraph (see Section 4.5 for complete listing).

4.3.4 Critical Appraisal Questions for Ecological Studies

Since ecology examines the interactions between living organisms and their environment and treats the ecosystem as a functional entity, ecotoxicology constitutes an interdisciplinary science. Therefore critical appraisal guidelines reflect, at least in part, those already discussed for epidemiology, analytic measurements and toxicology. Subject-specific elements need to be considered, such as the importance of indicating the level of observation (organism, population, or community), consistency of effects among different taxa, the comparison with historical data and the plausibility of other factors causing the observed effect. Further, are organisms employed as biomonitors to assess the presence of contaminants or as bioindicators to evaluate ecosystem health? The specific questions that are employed to address these quality of evidence issues are summarized in Section 4.5.

4.4 DESCRIPTION OF THE COMPUTERIZED INVENTORY OF LITERATURE ON LAKE ONTARIO

4.4.1 Introductory Comments

The purpose of the critical assessment was to devise a rapid, reliable and objective means for evaluating literature that is both extensive and diverse. Once summary information

is abstracted from a document or publication and entered into the database, it is essential to be able to manipulate it. A primary consideration in the development of the database is the production of new or alternative associations and relationships within the contents of a file or among different files.

4.4.2 Utility of the Database

Database use depends on the organization and the ease of recovery of information. Although each publication is categorized under one of the four headings, each database has the same structure facilitating searches for related data, impacts or impairments. Memos associated with each file may be viewed, printed or modified.

Publications may be evaluated using either a printed form (a sample is provided in Section 4.5), or by directly entering data into a screen-generated questionnaire. Included in the program format is the capability to generate displays which show the questions and responses given for any specific appraisal. This can be achieved from within dBASE or saved to a text file, which can be read or printed from a word processor employing appropriate software such as WordPerfect.

To achieve the goals outlined in Section 4.3 for critical appraisal, publications were assigned (in most cases) to one of four main categories:

A Toxic Effects *in vivo* or *in vitro*;
B Physical/Chemical or Analytical Data;
C Epidemiology; and
D Wider Ecological/Multifactorial Effects.

The objective of this classification was to facilitate the selection of questions relating to each subject area, and to keep the information organized into different database files. The quality of evidence questions address the five major appraisal elements outlined in Table 4.4.

Table 4.4. Appraisal elements that were considered.

i	Specific focus of the work (species, population, matrix)
ii	Outcome measured and methodology used
iii	Consistency and quality assurance
iv	Quantitative or qualitative aspects of exposure or effect
v	Statistical treatment of data

4.4.3 Structure of the Database System

The system is based on an application program of dBASE IV (copyright Ashton Tate, 1990). Each file consists of nineteen fields and a memo section. The title of each publication is catalogued in detail (Notebook II, version 3.0; Pro/Tem) and referenced by an accession number in the database. Information about each publication falls into three sections: *General Information, Impact and Focus, and Critical Appraisal.*

General Information. Information useful for the retrieval of a publication from the database is associated with each document reviewed. Questions include: date of publication; whether the source of information is primary (research paper), secondary (review article), or tertiary (interpretative by government or non-governmental organization). The location of the study site to which the research refers must be stated (e.g. nearshore/Toronto or offshore/East). A series of categorical queries (yes/no) to characterize the nature of the pollutant in question must be answered. Space has been provided for the inclusion of CAS# or other

identification number.

Impact. Under Annex 1 of the Great Lakes Water Quality Agreement (GLWQA), substances may be listed under three categories which establish a rough priority for level of concern. They are referred to in the database as *Primary Impact*. The GLWQA defines the three groups as in the following manner:

(i) List No.1 shall consist of all substances (1) believed to be present within the water, sediment or aquatic biota of the Great Lakes System and (2) believed, singly or in synergistic or additive combination with another substance, *to have acute or chronic toxic effects* on aquatic, animal or human life;

(ii) List No.2 shall consist of all substances (1) believed to be present within the water, sediment or aquatic biota of the Great Lakes System and (2) believed, singly or in synergistic or additive combination with another substance, *to have the potential to cause acute or chronic toxic effects* on aquatic, animal or human life; and

(iii) List No.3 shall consist of all substances (1) believed *to have the potential of being discharged* into the Great Lakes System and (2) believed singly or in synergistic or additive combination with another substance, to have acute or chronic toxic effects on aquatic, animal or human life.

Primary Focus. This designation determines which database file a publication falls under (i.e., category A, B, C or D; see Section 4.4.2). In the case of very large documents which address, for example, three of the four main areas for assessment, the publication can be subdivided and separate assessments given for each section. A simple change in the reference number (with attendant cross reference to the catalogue) suffices to keep the file unique. An opportunity to indicate a *Secondary Focus* of all publications is also included in the database. This is relevant since, for example, much of the toxicology literature reports both analytical and animal data, or contaminant content of herring gull eggs in combination with feeding behaviour of gulls.

Critical Appraisal. As already indicated, objectives on rules of evidence are tailored to the primary focus of a publication by devising separate questions for each primary data file. The complete set is listed in Section 4.5. Questions may be answered using key words or phrases (up to sixty characters). An essential point for the reviewer to remember is that these words are to be used to retrieve all related documents in the Inventory. Consistency in use of terminology is therefore important.

Finally, each document is categorized for the *Impairment of Beneficial Uses* explicitly mentioned in the publication under review.

At the conclusion of the appraisal section, a *Memo* is available which can include text such as comments about special strengths or weaknesses, or information about important references relating to the material presented in the publication (e.g., primary sources of data in the case of a review article). The *Memo* also provides an opportunity for a detailed

explanation of any of the appraisal points or subject material which the reviewer felt were particularly appropriate or not adequately covered by the appraisal format.

4.5 SAMPLE OF THE INVENTORY DATABASE

The flow diagram in Figure 4.1 depicts the various components and logistics of the database. Details of the *Data Entry* and *Critical Appraisal* schemes are also provided. It should be mentioned that early in the application of the critical appraisal questions, an "inter-rater reliability test" was conducted. Each of five team members made an independent assessment of three papers (a primary, secondary and tertiary source). Although there were some differences in assessment, particularly of the primary and secondary publications, these were readily resolved in a group discussion. The discrepancies were mostly concerned with distinguishing between the primary and secondary focus.

A limited number of printed copies of the complete database are available; upon request, the database program and textfile in an IBM-compatible-form can be provided.

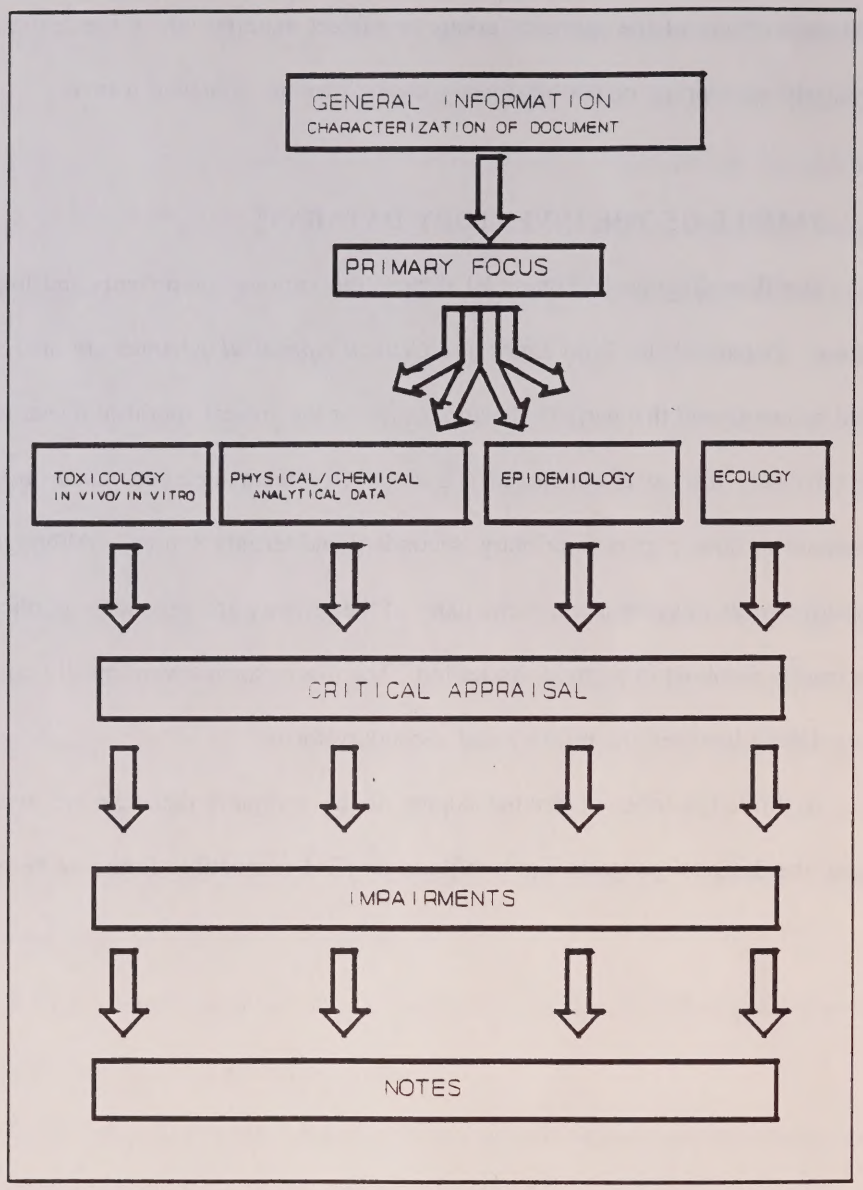

Figure 4.1. Flow diagram for major elements of the literature database. Each document is characterized by nineteen fields and a memo. Five fields are devoted to critical appraisal.

Title of Document:
Publication data:
Name of reviewer:
Version:22/5/91

DATA ENTRY FORM FOR LAMP STUDY

Reference Number: [ ] Date of Publication: [ ] Reference Type: [ ]

Describe the source of the information in the document.

PRIMARY (original paper published in a refereed journal)
SECONDARY (review/symposium proceedings refereed/unrefereed)
INSTITUTIONAL (Govt. document or product of industry, etc.)

Is it in the public domain? [ ]

If a specific LOCATION is addressed by the document, this should be indicated, otherwise indicate the general area of focus for the work described in the document:
[ ]

The Primary Pollutants discussed are:

Organic [ ] Physical [ ]
Inorganic [ ] Biological [ ]

Are the pollutants listed by the IJC? If so, what is the registration number (eg. CAS or IARC): [ ]

What is the PRIMARY IMPACT of the pollutant?

A: Responsible for identified acute/chronic toxic effect.
B: Has a potential for production of some toxic effect.
C: Toxicity unknown. Insufficient data re discharge or route of entry to system; perceived cause for concern. [ ]

The PRIMARY FOCUS of the DOCUMENT is:

A: Toxic effects *in vivo* or *in vitro*
B: Physical/chemical analytical data
C: Epidemiology
D: Wider Ecological/multifactorial effects [ ]

The SECONDARY FOCUS of the Document is:

A: Toxic effects *in vivo* or *in vitro*
B: Physical/chemical analytical data
C: Epidemiology
D: Wider Ecological/multifactorial effects [ ]

CRITICAL APPRAISAL OF DOCUMENT

Answer the following five questions with the PRIMARY FOCUS of the document regarding:

TOXIC EFFECTS *IN VIVO* OR *IN VITRO*

Appraisal Questions [A]

(1) What species were examined? Be specific.
(2) What outcome was measured? Mutagenicity, teratogenicity, tumor, organ dysfunction, metabolic dysfunction, etc.
(3) What is the Evidence for the effect? Autopsy, histology behaviour modification, physical deformity, etc.
(4) How was exposure assessed? Quantitative/qualitative; give details.
(5) Were the results analyzed for statistical significance? What method was used? Result/association?

Answer the following five questions with the PRIMARY FOCUS of the document regarding the:

ANALYSIS OF CHEMICAL OR PHYSICAL ATTRIBUTES OF POLLUTANTS

Appraisal Questions [B]

(1) What samples were analyzed? What/Where/When
(2) What was the analytical method(s) used? HPLC, GC, MS, enzyme function, immunity (eg. RIA, ELISA), chromosomal aberrations, etc.
(3) What is the level of QUALITY ASSURANCE? Were reference materials used (SRM), was a certified procedure used (eg. ASTM), previously cited?
(4) Was a dose/response relation developed? Was this quantitative? Details?
(5) Was the accumulation statistically significant relative to unpolluted samples or a specific source? What method was used? Confidence/result/association?

Answer the following five questions with the PRIMARY FOCUS of the document regarding the:

EPIDEMIOLOGY OF POLLUTION RELATED HEALTH EFFECTS

Appraisal Questions [C]

(1) What was the population studied? Workers, Indian Bands, mothers, children, general public? (include size).
(2) What was the STUDY DESIGN? Cohort, case-control, cross-sectional, etc.
(3) What was the follow up period? If retrospective give details.
(4) Was exposure assessment of the population quantitative/qualitative? Give details of measurement.
(5) Was the effect on the population STATISTICALLY SIGNIFICANT? What was the power of the tests employed?

Answer the following five questions with the PRIMARY FOCUS of the document regarding the:

ECOLOGICAL EFFECTS

Appraisal Questions [D]

(1) What level of ecosystem organization was studied (organism, population, community)?
(2) What ecological attribute was the focus of the study (e.g. behaviour, reproductive rates, predator-prey interactions, etc.)?
(3) Were the ecological confounding factors considered (e.g. age, sex, season, body weight, fat levels)?
(4) How was exposure assessed? Direct/Indirect effects.
(5) Was the ecological effect statistically significant relative to reference conditions?

IMPAIRMENTS OF BENEFICIAL USES

According to the IJC criteria, what uses are impaired?

(1) Restrictions on fish and wildlife consumption.
(2) Tainting of fish and wildlife flavour.
(3) Degradation of fish wildlife populations.
(4) Fish tumors and/or other deformities.
(5) Bird or animal deformities or reproduction problems.
(6) Degradation of benthos (bottom living plants or animals).
(7) Restrictions on dredging activities.
(8) Eutrophication or undesirable algae.
(9) Restrictions on drinking water consumption or taste and odor problems.
(10) Beach closings.
(11) Degradation of aesthetics.
(12) Added costs to agriculture or industry.
(13) Degradation of phytoplankton and zooplankton populations.
(14) Loss of fish and wildlife habitat.

NOTES:

CHAPTER 5 INVENTORY OF IMPAIRMENT OF BENEFICIAL USES IN LAKE ONTARIO

Annex 2 of the Great Lakes Water Quality Agreement as amended in 1987, outlines the general principles and framework for developing Remedial Action Plans (RAPs) in each of the 43 Areas of Concern (AOCs) and Lakewide Management Plans (LAMPs) for each of the Great Lakes. Annex 2 also specifies 14 ways in which beneficial uses can be impaired in the AOCs. These 14 impairments are the core of this study and the focus of Chapter 5.

Chapter 5 is divided into sections based on the use impairments in Annex 2. These sections are as follows:

5.1 Restrictions on fish and wildlife consumption;

5.2 Tainting of fish and wildlife flavor;

5.3 Degraded fish and wildlife populations;

5.4 Fish tumors or other deformities;

5.5 Bird or animal deformities or reproductive problems;

5.6 Degradation of benthos;

5.7 Restrictions on dredging activities;

5.8 Eutrophication or undesirable algae;

5.9 Restrictions on drinking water consumption or taste and odor problems;

5.10 Beach closings;

5.11 Degradation of aesthetics;

5.12 Added costs to agriculture and industry; and

5.13 Degradation of phytoplankton and zooplankton populations.

Each section of Chapter 5 consists of a definition of the use impairment followed by a description of where the impairment occurs in Lake Ontario, the causal pollutants that have been identified, and, if possible, the source of the impairment.

The fourteenth impairment listed in Annex 2 is "loss of fish and wildlife habitat". We have not included it here because the U.S. Environmental Protection Agency in cooperation with the U.S. Fish and Wildlife Service is planning a major habitat study in Lake Ontario in the near future.

Appendices are provided for several of the impairment sections (Sections 5.4, 5.5, 5.7, and 5.8). These appendices summarize the results of individual studies discussed in the relevant section (for Sections 5.4, 5.5, and 5.8) or location-specific data (Section 5.7).

A list of common and scientific names of the fish, birds, and animals discussed in the text is also provided in Appendix D.

5.1 RESTRICTIONS ON FISH AND WILDLIFE CONSUMPTION

5.1.1 DEFINITIONS

The International Joint Commission's (IJC) guidelines for listing and delisting impaired use 1, "restrictions on fish and wildlife consumption", are shown below.

Listing Guideline

When contaminant levels in fish and wildlife populations exceed current standards, objectives or guidelines, or public health advisories are in effect for human consumption of fish or wildlife. Contaminant levels in fish and wildlife must be due to contaminant input from the watershed.

Delisting Guideline

When contaminant levels in fish and wildlife populations do not exceed current standards, objectives or guidelines, and no public health advisories are in effect for human consumption of fish or wildlife. Contaminant levels in fish and wildlife must be due to contaminant input from the watershed.

The focus of these guidelines are the risks associated with consumption of contaminants to human health. The listing/delisting guidelines were formulated to account for differences in jurisdictional and federal standards and to emphasize local watershed sources. Thus a contaminant entering the Great Lakes primarily through atmospheric deposition, such as toxaphene from the southeastern United States, would not be considered sufficient to cause this use impairment.

Section 5.1 will focus on fish consumption advisories, particularly sport fish consumption advisories, because most of the available monitoring data involve fish. The major data sources are the Ontario Sport Fish Contaminant Monitoring Program (Ontario Ministries of Environment and Natural Resources, OME and OMNR, 1991) and the New

York Statewide Toxic Substances Monitoring Program (Sloan et al., 1987). Aside from official programs, Lake Ontario fish contaminant data are also available from other sources, such as the Canadian Department of Fisheries and Oceans Open Lake Fish Contaminants Surveillance Program (e.g., Borgmann and Whittle, 1991; Whittle and Keir, 1991). These data will be compiled and evaluated against the relevant jurisdictional criteria to supplement the information obtained from the official programs in New York and Ontario.

Published data from wildlife contaminant monitoring programs in New York and Ontario are nominal. There is some anecdotal data on waterfowl consumption advisories in the Niagara River area issued by New York and the latter jurisdiction has devised fish-flesh criteria for piscivorous wildlife, which will be discussed below. There is no Canadian program for wildlife consumption advisories. The Ontario Ministry of Natural Resources issued a press release in 1990 advising that the consumption of snapping turtles should be restricted due to contaminants in the flesh (V. Glooshenko, Ontario Ministry of Natural Resources, Toronto, personal communication).

5.1.2 CONSUMPTION ADVISORY GUIDELINES

Contamination of sport fish by toxic substances is a widespread problem in the Great Lakes (Baumann and Whittle, 1988; De Vault et al., 1989). Ontario and New York have monitored contaminant levels in Lake Ontario sport fish and issued consumption advisories since the early 1970s and mid-1970s respectively (Armstrong and Sloan, 1980; OME and OMNR, 1991). Consumption advisories are based on tissue concentrations of contaminants in different size classes and species of fish and the relationship between length (age), tissue

concentrations, and trigger levels. The current contaminant guidelines or trigger levels used by New York and Ontario are shown in Table 5.1.1. These guidelines were developed using a risk assessment approach to calculate exposure limits that are associated with negligible lifetime cancer risk and are considered to protect public health (Cox and Johnson, 1990). When tissue concentrations exceed these levels, they trigger the issuing of consumption advisories ranging from restrictions of 2.3 kg per week to do-not-consume.

Guidelines from the Great Lakes Water Quality Agreement (GLWQA) are also shown in Table 5.1.1. The GLWQA guidelines were developed to provide protection to aquatic organisms and fish-consuming birds and they relate to whole-fish concentrations rather than concentrations in fillets. From an ecosystem perspective, whole-fish levels might be most appropriate. The GLWQA guidelines for PCBs, mirex and DDT are substantially more stringent than those used by Ontario and New York for the protection of human health, but neither Ontario nor New York offer enforceable protection for the GLWQA guidelines.

There are substantial differences in advisory development procedures in Ontario and New York, as well as statutory and philosophical differences with respect to the issuance of fish consumption advisories. In Ontario, contaminant data for a species are used to calculate a regression of tissue concentration against length for individual contaminants; these regressions are subsequently used to estimate the length at which the trigger level is exceeded in each species. Advisories restricting consumption are then issued for fish in this size class and above. If more than one contaminant exceeds the relevant trigger level, then the advisories are based on the contaminant which exceeds the trigger level at the lower length. Information on contaminants, species and lengths, and location is provided. Ontario advises

Table 5.1.1. Concentrations of contaminants which trigger consumption advisories in Lake Ontario sport fish. Units are ppm (μg/g, wet weight) unless otherwise indicated. Blank spaces indicate no trigger level was found. NE - none established.

Parameter	Ontario[a]	New York[b]	GLWQA
Sample portion	muscle fillet	side fillet	whole fish
2,3,7,8-TCDD (ppt)	20.0	10.0	
DDT and metabolites	5.0	5.0	1.0
mirex	0.1	0.1	0[d]
PCB	2.0	2.0	0.1
mercury	0.5	1.0[c]	0.5
chlordane	0.3[f]	0.3	
toxaphene	3.0[g]		
lindane	0.1[g]	NE	0.3[h]
aldrin	0.1[g]	0.3	0.3[e,h]
heptachlor	0.1[g]	0.3	0.3[h]
dieldrin	0.1[g]	0.3	0.3[e,h]
endrin	0.1[g]	0.3	0.3[h]
copper	2.0[g]		
zinc	10.0[g]		
lead	1.0[g]		
arsenic	0.3[g]		

[a] Guidelines for TCDD, DDT, mirex, PCB and mercury from Cox and Ralston (1990).
[b] Guidelines from Sloan et al. (1987).
[c] Measured as methylmercury.
[d] Actual criterion is substantially absent, meaning below detection limits.
[e] The sum of dieldrin and aldrin concentrations should not exceed 0.3 μg/g, wet weight.
[f] U.S. FDA figure as no Canadian guideline is available.
[g] Suggested, but not official guidelines in Ontario. Consumption advisories have not been triggered by these substances.
[h] Concentrations for edible portion (fillet) not whole fish.

restricting consumption of fish contaminated with of one or more industrial organic contaminants, such as PCBs or mirex, at levels above the guidelines to 1-2 meals per week or month depending on the frequency of consumption (OME and OMNR, 1991).

Consumption of mercury contaminated fish is initially restricted at mercury levels between 0.5 and 1.0 ppm and is further restricted when levels are between 1.0 and 1.5 ppm. If mercury levels exceed 1.5 ppm, then humans are advised not to consume these fish. Women of childbearing age and children under 15 are advised to consume only fish with no consumption restrictions due to mercury or organic compounds. Fish for which consumption is not restricted are not necessarily free of contaminants; rather tissue concentrations are below the trigger levels in Table 5.1.1. A substance is legally a "contaminant" in fish flesh in Ontario when it is found there at levels greater than the relevant criterion. When all substance tested are below their trigger levels, Ontario issues an advisory that consumption is not restricted.

The method used by New York to develop consumption advisories is based on the levels shown in Table 5.1.1 and average tissue concentrations of contaminants (Foran and VanderPloeg, 1989). The ratio of tissue concentration and trigger levels is summed for all organic contaminants (F). When $F < 1.5$ New York advises that no more than one meal per week be consumed, when $1.5 \leq F < 3$ consumers are advised to consume fish no more than once per month, and when $F \geq 3$ anglers are advised not to consume fish (Foran and VanderPloeg, 1989). Unlike Ontario, these advisories are issued for all locations in Lake Ontario and the Niagara River below the falls rather than for specific locations. The New York approach also differs from Ontario's in that it is based on the entire suite of organic contaminants in a fish rather than individual contaminants. However, New York does not incorporate mercury into its addivity formula and it does not appear to account for the effect of age on bioaccumulation in fish as in Ontario's approach. Under New York's system

unrestricted consumption would occur only when fish are below the detection limits for the substances tested. However, in practice there are no freshwater fish species in New York for which unrestricted consumption is advised as the state has issued a blanket general advisory recommending against such practice.

Comparison of the fish contaminant data and subsequent consumption advisories is further hampered by the use of different tissue samples in contaminant analysis. Ontario uses a lean, boneless, skinless fillet of dorsal muscle between the dorsal and pectoral fins and above the lateral line (Cox and Ralston, 1990). The rationale for this sample is that it provides the most consistent test results and it is the best edible portion of most fish. When organic contaminants such as PCBs are a concern, Ontario advises filleting and trimming which results in an edible portion similar to the analyzed portion. In contrast, New York uses a standard fillet consisting of an entire side of a fish from just behind the operculum to the tail and including skin, bones of the half the rib cage and one pelvic fin, but excluding the vertebral column, internal organs, dorsal, pectoral, anal, and caudal fins (Armstrong and Sloan, 1987). New York advises, however, that all excess fat and skin should be removed by consumers which results in an edible portion that is different than the analyzed portion on which the consumption advice was based.

The question of a standardized portion type has been the subject of discussions between the Great Lakes states and Ontario since 1986 (Great Lakes Water Quality Board, GLWQB, 1989c). Comparative data on fish preparation techniques noted fairly consistent ratios for PCB and total DDT concentrations in Lake Ontario fish of 1.0:0.8:0.5 for whole fish:U.S. fillet:Canadian fillet (L. Skinner, cited in GLWQB, 1989b). The Canadian fillet

contains significantly less lipid than the U.S. fillet or whole fish and as a result accumulates lower levels of lipophilic contaminants. The analyses for mirex and HCB were incomplete in that only 2 of 3 preparations were tested, but the concentrations were also generally consistent with the PCB/DDT ratio. Mercury was apparently unaffected by the preparation technique used, probably because it is not lipophilic. Recent work (1990) in Ontario indicates that lipid levels and PCB and mirex concentrations in skinless and skin-on fillets of coho and chinook salmon are much lower than in the mid-1980s and are the same in each species (A. Johnson, Ontario Ministry of the Environment, Toronto, Ontario, unpublished data). As a result, there is little difference in the advisories which would be issued using either fillet type for these species.

Since the final decision concerning consumption is left to the consumer, regardless of the advisories, neither analytical sampling will cover all contigencies. Ontario anglers utilizing other portions of the fish, for example, whole gutted fish, fish steaks, or belly fat, are advised that they will probably encounter higher contaminant concentrations than indicated by the test results. The use of the whole fillet in New York may over estimate human uptake. For example, fishermen in New York generally trim fatty material and are consuming portions of salmon in which mirex concentrations are, depending on weight, near or below the trigger level (Insalaco et al., 1982). Selectively removing and trimming the red muscle, belly flap, and skin from the fillet results in at least a 56% reduction in mirex in the consumable portion of salmon. Skea et al. (1979) reported that trimming smallmouth bass and brown trout from Lake Ontario to remove skin, dorsal and ventral fat, and the lateral line caused the largest reductions of mirex, PCB, and DDE (43-64%).

The consumption advisories issued by Ontario and New York are updated as new analytical data are compiled. However, not all sites are resampled annually, thus consumption advisories for different locations, and even for different species within a location are not necessarily based on concurrent data; in fact they may consist of data from the early 1980s to 1990. This is particularly true in Ontario where consumption advice is provided for more than 1,600 locations across the province and resources must be allocated between testing new locations and retesting those locations where changes in contaminant levels may be occurring (OME and OMNR, 1991). Coho salmon have been collected at the mouth of the Credit River annually since 1972 and contaminants have been analyzed most years except 1973 and 1980. This probably respresents the longest contaminant data series for a particular species in Lake Ontario. We have not attempted to determine the age of the data, but we note that most species in most locations in Lake Ontario have been retested within the last four years, particularly the salmonines. The Ontario monitoring program on Lake Ontario is much larger in terms of the number of locations and species tested than the New York program which is more modest.

5.1.3 LAKE ONTARIO FISH CONSUMPTION ADVISORIES

Advice that fish consumption should be restricted was issued in 1990 for 21 species from 35 locations in Lake Ontario by the Province of Ontario (Table 5.1.2). These data do not include fish consumption advisories in which consumption is not restricted (Ontario only). All of the Ontario AOCs list restrictions on fish consumption as a use impairment. Mirex, PCBs, 2,3,7,8-TCDD, and mercury were the only contaminants leading to restrictions in Lake

Table 5.1.2. Locations and contaminants triggering advisories restricting the consumption of fish from Lake Ontario. Ontario data are listed from west to east. Advisories in which consumption is not restricted are not shown.

Location	Fish species	Contaminant
	Ontario[a]	
Upper Niagara River		
Fort Erie	freshwater drum	PCB
Miller Creek	white sucker	mercury
Lower Niagara River		
Queenston-Whirlpool	American eel	PCB
	rainbow trout	mirex
	yellow perch	mercury
	coho salmon	mirex
	smallmouth bass	mercury
	white perch	PCB
	lake trout	mirex
	channel catfish	PCB
	freshwater drum	mercury
	chinook salmon	mirex
Lake Ontario #1		
Jordan Harbour to Port Weller	brown trout	mirex
	channel catfish	mercury
	lake trout	2,3,7,8-TCDD
	freshwater drum	mercury
	carp	PCB
	chinook salmon	mirex
Burlington Bay, Hamilton Harbour	white perch	mercury
	carp	PCB
	channel catfish	PCB
	brown trout	mirex
	freshwater drum	mercury
Bronte Creek	chinook salmon	mirex
	coho salmon	mirex
	brown trout	PCB
Credit River	chinook salmon	mirex
	coho salmon	mirex

Table 5.1.2. Continued.

Location	Fish species	Contaminant
Credit River	rainbow trout	mirex
	lake trout	2,3,7,8-TCDD
	brown trout	2,3,7,8-TCDD
Long Branch	lake trout	PCB
Humber River mouth	brown trout	mirex
Humber Bay area	lake trout	PCB
Hearn GS - Outer	carp	mirex
harbour	white bass	mercury
	yellow perch	mercury
	gizzard shad	PCB
	northern pike	mercury
Toronto Islands,	white sucker	mercury
Inner harbour	northern pike	mercury
	carp	PCB
Ashbridges Bay	brown trout	mirex
Scarborough Bluffs	lake trout	mirex
	brown trout	mirex
Lake Ontario #2		
Frenchman's Bay	brown bullhead	mirex
	carp	mirex
Pickering NGS	rainbow smelt	mirex
	brown trout	PCB
	white bass	mercury
	walleye	mercury
Oshawa area	rainbow trout	mirex
	coho salmon	mirex
	chinook salmon	mirex
Ganaraska River	rainbow trout	mirex
	brown trout	mirex
	lake trout	PCB
	chinook salmon	mirex
Lake Ontario #3		
Presqu'ile Bay	largemouth bass	mercury
Scotch Bonnet	lake trout	mirex

Table 5.1.2. Continued.

Location	Fish species	Contaminant
Gravelly Bay	walleye	mercury
Lake Ontario #4 - Bay of Quinte		
General	channel catfish	mercury
Trent River mouth	walleye	mercury
Trenton to Belleville	walleye	mercury
Belleville, Telegraph Narrows, Long Reach	walleye	mercury
Big Bay	walleye	mercury
Hay Bay	walleye	mercury
Keith Shoal, Adolphus Reach	walleye	mercury
Glenora to Upper Gap	American eel	PCB
Lake Ontario #6		
Prince Edward Bay, Long Point	American eel	PCB
Nearshore, North channel	walleye	mercury
	northern pike	mercury
	smallmouth bass	mercury
	chinook salmon	mirex
Main Duck Island	American eel	PCB
	lake trout	mirex
	smallmouth bass	mercury
	brown trout	mirex
Lower Gap	lake whitefish	mirex, PCB
	northern pike	mercury
Reeds Bay	American eel	PCB
New York		
Lake Ontario[b]	chinook salmon[c]	dioxin, mirex
	lake trout[c]	dioxin, PCB, mirex
	coho salmon[c]	mirex
	rainbow trout[c]	mirex

Table 5.1.2. Continued.

Location	Fish species	Contaminant
Lake Ontario[b]	brown trout[c]	PCB, mirex
Black River Bay	American eel[d]	PCB, mirex
Hardscrabble	American eel[d]	PCB, mirex
East of Oswego Harbour	American eel[d]	PCB, mirex
Eastern Basin	white perch[e]	PCB, mirex
Eastern Basin[f]	channel catfish[d]	PCB, mirex

[a] Information on organic substances causing consumption restrictions was obtained from C. Cox, Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario.

[b] Precise locations not specified but probably Salmon River, Rochester, and Sandy Creek for the salmons, brown and rainbow trout. Lake trout likely sampled at Pultneyville and Stony Island.

[c] Data from 1989 fall collections except brown trout data, which is taken from fall 1990 collections. All data are unpublished from Skinner 1991.

[d] Mean tissue concentrations in 1983 collections reported by Sloan et al., (1987).

[e] Mean tissue concentrations in 1984 collections reported by Sloan et al., (1987).

[f] Mean tissue levels in channel catfish collected from Chaumont, Black River, and Henderson Bays in 1983.

Ontario localities. Other organic substances such as DDT and its metabolites, lindane, heptachlor, aldrin, and dieldrin as well as heavy metals including copper, zinc, cadmium, nickel, lead, and arsenic have been tested at various locations and were found to be lower than the proposed trigger levels for these substances in Table 5.1.1. Of the 174 listings of species tested at specific locations in Lake Ontario, 92 indicated that all sizes of that species were below the trigger levels of the substances tested. Hence anglers were advised that consumption was not restricted. Some degree of restriction was advised for 81 listings, of which 69 had smaller sizes of the particular species listed for which consumption was not

restricted and 12 had restrictions on consumption of all sizes of a species tested. Only contaminants causing restrictions are shown in Table 5.1.2; others, such as PCBs, may exceed their trigger levels in the same species and location, but at higher lengths.

Ontario has restricted the consumption of lake trout at Jordan Harbour and lake and brown trout at the Credit River based on test results for the most toxic dioxin isomer, 2,3,7,8-TCDD. The incorporation of isomer-specific analyses for other dioxins and dibenzofurans, expressed as toxic equivalents of 2,3,7,8-TCDD (Safe, 1990), is being considered. If this approach is taken, then consumption restrictions also would be issued for all listed sizes of lake trout at Jordan Harbour (>35 cm), and lake trout (>35 cm) and brown trout (>45 cm) at the mouth of the Credit River (OME and OMNR, 1991).

The New York State Department of Health (NYSDH, 1990) has a blanket general advisory which recommends that individuals should "... eat no more than one meal (one-half pound) per week of fish from the State's freshwaters...". This advisory is designed to protect consumers against the ingestion of fish from water bodies that have yet to be tested or which may contain unidentified contaminants. The New York data in Table 5.1.2 (discussed below) relate to advisories for specific waterbodies in which monitoring has revealed contaminants that exceed the relevant action levels.

The New York data specific to Lake Ontario are not easily compared with the Ontario data. The individual contaminant data were evaluated against their criterion to compile the list in Table 5.1.2 and eight fish species from at least 12 locations had levels of contaminants high enough to trigger consumption advisories. The NYSDH (1990) recommends not eating Amercian eel, channel catfish, carp, lake trout, chinook salmon, coho salmon over 53 cm,

rainbow trout over 63.5 cm, and brown trout over 51 cm captured in all New York waters of Lake Ontario below Niagara Falls. Consumption of white sucker, white perch, and smaller coho salmon, rainbow and brown trout is restricted to no more than one meal (0.25 kg) per month. Women of childbearing age and children under 15 are advised not to consume fish with elevated contaminant levels, that is, any fish for which there are consumption advisories.

Consumption advisories due to PCBs and mirex predominated in the western basin of Lake Ontario whereas restrictions due to mercury were most common in the eastern basin, particularly in the Bay of Quinte. These differences are not related to selective testing of substances, since recent retesting throughout the western basin has included organic and inorganic substances. Mirex was responsible for triggering more consumption advisories than either PCBs or mercury (Table 5.1.3). Mercury was found only in cool- and warmwater species whereas PCBs and mirex predominated in the salmonines. Cool- and warmwater fishes tend to inhabit shallow, nearshore areas and are less mobile than the salmonines which are usually found in the deeper offshore waters. Cool and warmwater fishes predominate in the eastern basin where there is abundant habitat and coldwater fish are more common in the fisheries of the central and western basins of Lake Ontario. These results suggest that mercury contamination in Lake Ontario is primarily a nearshore problem; PCBs, mirex, and possibly dioxins are a more widespread problem. Mirex currently appears to be the more limiting contaminant because it causes restrictions in smaller sized fish than either PCBs or mercury. The mercury data from New York is not sufficient to judge the nature and extent of the contaminantion in New York waters of Lake Ontario.

Table 5.1.3. Contaminants causing fish consumption restrictions in Lake Ontario species in 1990. Data are number of locations reporting restrictions for a given species from the Ontario data in Table 5.1.2. A blank space indicates that no advisories restricting consumption were reported.

Fish species	PCB	Mirex	Mercury	2,3,7,8-TCDD
rainbow trout		4		
lake trout	3	4		2
brown trout	2	7		1
coho salmon		4		
chinook salmon		7		
lake whitefish	1			
rainbow smelt		1		
freshwater drum	1		3	
white sucker			2	
American eel	5			
walleye			9	
yellow perch			2	
smallmouth bass			3	
largemouth bass			1	
white perch	1		1	
white bass			2	
brown bullhead		1		
channel catfish	2		2	
carp	3	2		
gizzard shad	1			
northern pike			4	
Total	19	31	29	3

Assessments of the level of awareness concerning fish consumption advisories, particularly those recommending restricted consumption, have been targeted at fish consuming segments of the populations of New York and Ontario. The only study to evaluate consumer awareness in New York indicated that fish-consuming, low-income women

living near Lake Ontario knew little about the existence of the guidelines and that there was a general misunderstanding of the potential effects of contaminants in the diet (Wendt (1986), cited in Brown et al., 1991). Cox and Johnson (1990) surveyed Ontario anglers on Lake Ontario, Lake Huron, and Georgian Bay and reported that where there were consumption restrictions on salmon and trout (Lake Ontario), anglers consumed less of these species than did anglers fishing Lake Huron or Georgian Bay where there are few or no restrictions. Almost 93% of Lake Ontario anglers were aware of the consumption guidelines and this combined with their species preferences for consumption – less than 10% consumed lake trout which usually has the highest contaminant concentrations and more than 90% consumed rainbow trout which is not restricted at smaller sizes – follow the consumption advisories. Unfortunately there is no data for other potential consumer groups.

5.1.4 FISH CONTAMINANT DATA FROM SURVEILLANCE PROGRAMS

Surveillance programs monitoring a large array of toxic substances in fish have been carried out by federal, state, and provincial agencies, to monitor temporal and spatial trends in contaminants. The data for Lake Ontario will be briefly summarized here; for a more comprehensive discussion see Baumann and Whittle (1988), Johnson et al. (1989), Government of Canada (1991a), Borgmann and Whittle (1991), and Suns et al. (1991). Concentrations of many contaminants in fish decreased between the early 1970s and the late 1980s. The declines were greatest through the 1970s for most contaminants, with a slowing in the rate of reduction in the 1980s. These sharp initial reductions probably reflect improved industrial practices, more stringent regulations, and bans or restrictions on the manufacture

and use of many substances. Although there is considerable variation in the data, the mean concentration of PCB in coho salmon from the Credit River declined exponentially between 1972 and 1990 (A. Johnson, unpublished data). PCB levels reached a low of approximately 1.0 ppm in 1988 and have risen since this time. Mean PCB levels are currently near 2.0 ppm (OME and OMNR, 1991), which is lower than when monitoring began, but is higher than in fish from the other Great Lakes (Government of Canada, 1991a). Mirex concentrations have declined slightly between 1976 and 1986, but in 1987 and 1988 mirex levels in coho salmon from the Credit River were below the 0.1 ppm criterion (A. Johnson, unpublished data). Current levels are sufficient to cause the majority of fish consumption restrictions in Lake Ontario. Similarly, mercury levels have dropped since the late 1970s-early 1980s, but mercury continues to cause restrictions in nearshore species, particularly in the eastern basin (Government of Canada, 1991a).

Data from some of the surveillance programs were evaluated against the trigger criteria in Table 5.1.1 and those which exceed the individual levels are shown in Table 5.1.4. Most of these programs report concentrations on a whole fish basis. These data were converted to an edible portion basis (Ontario fillet) using the ratio 1.0:0.5 (whole fish:Ontario fillet) so that the data are comparable with those used to compile Table 5.1.2. The results are consistent with the analysis of the fish consumption advisory data: PCBs, mirex, and 2,3,7,8-TCDD were the only contaminants at sufficiently high concentrations to trigger consumption restrictions. Many of these studies also reported the results of analyses for a suite of substances for which consumption limits have not been established yet.

Table 5.1.4. Mean organic contaminant concentrations exceeding consumption guidelines in Lake Ontario fish from other studies. Data were originally reported on a whole fish basis and have been converted to edible portions using the ratio 1.0:0.5 (whole fish:Ontario fillet). Concentrations are in $\mu g/g$, wet weight unless indicated otherwise.

Location	Date	Fish species	Weight (kg)	Contaminant (Concentration in edible portion)	Reference
Hamlin and Oswego	1984	lake trout[a]	2.86	2,3,7,8-TCDD (24.5 ppt)	1
				PCB (2.7)	
Port Credit	1987	lake trout[c]	2.41	PCB (4.98)	2
				mirex (0.22)	
Credit River	1987	rainbow trout[d]	3.38	PCB (2.83)	2
				mirex (0.12)	
	1987	coho salmon[d]	3.33	PCB (2.32)	2
				mirex (0.10)	
Fort Niagara	1984	carp[e]	3.2	mirex (0.29)	3
				PCB (12.6)	3
12 Mile Creek	1984	carp[e]	3.6	mirex (0.37)	3
				PCB (13.6)	3
18 Mile Creek	1984	carp[e]	3.1	mirex (0.7)	3
	1984	goldfish[e]	0.5	mirex (0.28)	3
Oak Orchard	1984	catfish[f]	0.7	mirex (0.20)	3
Sodus Bay	1984	carp[e]	4.7	mirex (0.28)	3
Black River	1984	white sucker	0.7	mirex (0.15)	3

References: 1 - De Vault et al., (1989); 2 - Niimi and Oliver, (1988); 3 - Jaffe and Hites.
[a] Five composite samples of five whole fish from each location.
[b] Whole lake trout collected in 1985
[c] Whole fish analysis on a sample of 10 fish.
[d] Whole fish analysis based on a sample of 12 fish.
[e] Composite whole fish analysis (N=3).
[f] Based on sample of 4 fish.

The Canada Department of Fisheries and Oceans has monitored contaminant levels in Lake Ontario lake trout from six locations since 1977 as part of the Open Lake Fish Contaminants Surveillance Program. The analytical results are reported on a whole fish basis using age 4+ fish as a reference age so that annual information on trend analysis is based on a consistent exposure period (e.g., Baumann and Whittle, 1988; Whittle and Keir, 1991). Since these data were not originally collected for consumption advisory purposes, the data expressed on a whole fish basis have again been converted to edible portions using the ratio 1.0:0.5 (whole fish:Ontario fillet) for comparability with the data compiled from the Ontario Sport Fish Contaminant Monitoring Program.

Contaminant concentrations in age 4+ lake trout have declined over the 1977 to 1989 period as have body weight and lipid content (Table 5.1.5). Mercury and dieldrin levels have declined steadily over the study period whereas there has been much year-to-year variation about the general declining trend of mirex in lake trout tissues. DDT concentrations declined steadily until 1986, but have increased consistently in the 1987-1989 period. In contrast, PCBs decreased between 1977 and 1981, increased from 1982 to 1984, declined in 1985, peaked again in 1987, and are now somewhat lower. Current contaminant levels in age 4+ lake trout are such that an advisory restricting consumption would be issued for mirex. The other contaminants have concentrations below their action levels.

The decline in lipid concentrations in lake trout between 1977 and 1989 complicates the interpretation of temporal trends in lipid soluble organic contaminants. Borgmann and Whittle (1991) developed a regression model to assess the trend in time and the effects of age, sampling site, and weight on concentrations of the substances in Table 5.1.5 in lake trout

Table 5.1.5. Mean contaminant concentrations in age 4+ lake trout from Lake Ontario, 1977 to 1989. Organochlorine data were converted from whole fish to edible portion using the ratio 1.0:0.5 (whole fish:Ontario fillet). Mercury data are as reported. All data were taken from Whittle and Keir (1991). Concentrations are reported as μg/g, wet weight unless otherwise indicated.

Year	N	Weight (g)	Lipid (%)	Contaminant				
				PCB	DDT	Dieldrin	Mirex	Mercury
1977	3	2102.3	21.30	4.00	2.18	-	0.245	0.25
1978	11	1199.5	17.75	2.53	0.64	0.105	0.075	0.18
1979	75	2067.8	20.23	2.36	1.50	0.115	0.125	0.20
1980	82	1791.7	18.42	2.38	0.40	0.06	0.07	0.20
1981	83	1538.0	19.09	1.84	0.76	0.055	0.075	0.16
1982	36	1861.7	19.25	2.94	0.56	0.075	0.08	0.17
1983	46	1769.1	18.21	3.22	0.78	0.07	0.105	0.22
1984	47	1546.9	17.39	2.96	0.66	0.075	0.04	0.14
1985	14	1713.6	15.08	1.40	0.44	0.055	0.065	0.14
1986	25	1720.9	16.70	1.44	0.44	0.07	0.03	0.11
1987	64	1826.0	15.65	1.74	0.47	0.065	0.055	0.13
1988	46	1771.4	14.83	1.70	0.60	0.06	0.10	0.11
1989	34	1675.2	14.05	1.61	0.68	0.04	0.10	-

using the entire data set collected by DFO (age 1+ to 8+ fish from six sites). They found that lipid normalized contaminant concentrations within an age class (e.g., 4+ fish) and lipid normalized mirex, chlordane, dieldrin, and mercury concentrations across age classes, decreased with increasing body size. But the body size effect was less within an age class than across age classes. Citing food chain biomagnification as the primary determinant of contaminant concentrations in fish, Borgmann and Whittle (1991) suggested that the fish's ability to excrete a contaminant relative to the rate of increase in lipid concentration with body size is a major factor controlling contaminant concentrations in lake trout from Lake Ontario. Thus the decline in lipid soluble organic contaminants in 4+ lake trout during the 1977-1989 period may be related to changes in body size as well as load reductions to Lake Ontario.

Whittle and Keir (1991) also reported 2,3,7,8-TCDD residue concentrations (whole fish) in lake trout collected from Lake Ontario between 1977 and 1989. These data are derived from direct measurements in tissues since 1986 and from retrospective analyses of archived tissue samples dating from 1977. There is considerable variation in these data, in part because the number of samples is limited; most annual mean concentrations were greater that 30 ppt except in 1977 and 1984 when concentrations were 10-15 ppt. The mean concentration of 2,3,7,8-TCDD has increased significantly between 1981 and 1988 (SNK multiple range test, $P < 0.05$). The mean TCDD concentration declined in 1989 but was still an order of magnitude greater than concentrations measured in lake trout from the other Great Lakes.

PCB, mirex, and DDT residue data from nearshore collections of young-of-the-year spottail shiners are shown in Table 5.1.6. Residues of HCB, heptachlor, aldrin, total hexachlorocyclohexane, octachlorostyrene, total chlordane, and toxaphene were generally near or below their detection limits in Ontario shiner collections (Suns et al., 1991) and consequently are not included. PCB and total DDT residues in recent shiner collections were significantly lower ($P < 0.01$) relative to levels in the 1970s, but the rate of decline has slowed during the 1980s (Suns et al., 1991). None of the spottail shiner contaminant residues approach levels triggering consumption restrictions in the sport fish monitoring programs. Young-of-the-year fish are not normally consumed by humans. However, total PCBs at 15 sites and mirex at 8 sites in Lake Ontario exceed the criteria recommended for the protection of fish consuming wildlife by Newell et al. (1987) (Table 5.1.7) and the GLWQA criteria for protecting aquatic life (Table 5.1.1). The criteria for these contaminants are more stringent than those used in either of the adult fish contaminant monitoring programs, but they are not enforceable. The guidelines developed by Newell et al. (1987) are designed to protect piscivorous wildlife from the carcinogenic effects of organic substances. If young-of-the-year shiners were the sole food source of piscivores, then these data would be indicators of potential human uptake of contaminants (Skinner and Jackling, 1989).

Collectively, the data from the young-of-the-year shiner and other surveillance programs indicate that PCBs, mirex, and 2,3,7,8-TCDD are the causal pollutants restricting fish consumption in Lake Ontario. Because young-of-the-year spottail shiners are restricted to nearshore habitat and have undifferentiated food habits, contaminant residue data reflect site-specific availability from water borne sources (Suns et al., 1991). These data suggest that

Table 5.1.6. Organochlorine concentrations (ng/g) in young-of-the-year spottail shiners from Lake Ontario. Data are from 1987 unless otherwise indicated and consist of means (standard deviation). ND - no data, TR - trace concentration, not quantified.

Location	N[a]	Fat (%)	PCB	DDT	Mirex
Lewiston[b]	7	1.7(0.4)	99(36)	14(11)	ND
Peggys Eddy	6	2.2	133(92)	23	ND
Niagara-on-the-Lake[b]	7	2.4(0.4)	145(59)	47(30)	ND
Welland Canal	7	1.8(0.2)	127(17)	39(9)	14(2)
Twelve Mile Creek	7	2.2(0.2)	207(36)	85(13)	12(2)
Burlington Beach	5	2.6(0.2)	194(20)	47(6)	7(3)
Hamilton Harbour	6	5.0(0.4)	338(30)	33(30)	TR
Cootes Paradise	6	3.8(0.3)	368(54)	48(21)	ND
Bronte Creek	7	3.0(0.3)	171(27)	36(13)	12(3)
Oakville Creek	7	2.4(0.2)	101(19)	28(6)	ND
Credit River[b]	7	4.0(0.6)	264(48)	80(55)	ND
Mimico Creek	7	4.9(0.5)	173(30)	35(7)	ND
Humber River	6	5.2(0.6)	235(33)	31(4)	ND
Toronto Harbour	5	2.7(0.4)	132(25)	20(2)	ND
Leslie Spit	4	3.4(0.3)	185(18)	20(3)	ND
Bluffers Park	5	3.4(0.3)	ND	6(2)	ND
Rouge River	5	2.7(1.0)	78(17)	26(6)	ND
Napanee River[b]	5	2.2(0.2)	162(51)	9(5)	ND
Outlet Park[c]	7	3.3(0.5)	26(17)	9(4)	ND
Wolfe Island	7	2.5(0.5)	33(17)	7(5)	24(3)
Genesee River[d]	10	3.9(0.5)	199	54(9)[e]	3(1)
Oswego Harbour	10	1.1(0.2)	38(8)	8(2)[e]	2(0.5)
Salmon River	10	1.2(0.1)	52(11)	12(2)[e]	2(3)
Black River Bay	10	1.6(0.2)	313	4(2)[e]	ND

Data sources: Ontario - Suns et al. (1991); New York - Skinner and Jackling (1989).
[a] Composite samples of 10 fish each.
[b] Collections made in 1988.
[c] Collections made in 1986.
[d] Emerald shiners analyzed.
[e] Data for p,p'-DDE.

Table 5.1.7. Recommended guidelines for contaminant residues in fish for the protection of piscivorous wildlife in the GLWQA and developed by Newell et al., (1987) for the NYDEC. Data are in ppb (ng/g).

Contaminant	Newell et al. criteria	GLWQA criteria
Total PCB	110	100
aldrin/dieldrin	120	
Total DDT	200	
chlordane	500	
dioxin (2,3,7,8-TCDD)	0.003	
endrin	25	
HCB	330	
heptachlor and heptachlor epoxide	200	
hexachlorocyclohexane (including lindane)	100	
mirex	330	0[a]
octachlorstyrene	20	

[a] The GLWQA criterion requires that mirex to be virtually absent in biota for the protection of wildlife.

the western end of Lake Ontario and the Niagara River are major sources of organochlorine contaminants in fish. The site-to-site variations in PCB, DDT, and mirex in lake trout collected by DFO also suggest a source(s) for these substances in the western end of Lake Ontario (Borgmann and Whittle, 1991).

5.1.5 WATERFOWL

The NYSDH (1990) recommends that hunters not consume mergansers as they are the most heavily contaminated waterfowl tested. Other waterfowl should be skinned and all fat

removed and consumption limited to two meals per month. The monitoring data on which these recommendations were based also indicate that wood ducks and Canada geese are less contaminated than other waterfowl species, with dabbler ducks and then diving ducks having increasingly higher contaminant concentrations. These advisories are not specific to Lake Ontario; rather they appear to be state-wide. The data on which these recommendations were made were not available.

Organochlorine contaminants in common goldeneye wintering on the upper Niagara River, New York, were measured by Foley and Batcheller (1988). Hatching-year (< 1 yr old) and adult (> 1.5 yr old) males were collected near their time of arrival on wintering grounds (November-December) and adult males just prior to dispersal to breeding grounds (February-March) to identify the contaminants that occur in these birds and to compare changes in concentrations in birds from early and late winter. Mean concentrations of all measured contaminants were significantly greater ($p < 0.01$) in adult birds than in hatching-year birds collected in early winter (Table 5.1.8). Adult birds had significantly greater ($p < 0.01$) concentrations of all contaminants, except mirex and oxychlordane ($p > 0.05$), at the end of their stay than they did near their time of arrival. The data for hatching-year birds indicate that common goldeneye ducks accumulate contaminants prior to their appearance on the wintering grounds while the comparison of adult birds demonstrates that common goldeneye also accumulate significant quantities of contaminants during the relatively shore over-wintering period, likely from food sources. Foley and Batcheller (1988) assumed that the age distribution (these ducks live to about 6 years but age classes > 1.5 years cannot be reliably distinguished) of the ducks did not change over the winter and that the same population was

Table 5.1.8. Mean organochlorine concentrations (ppm, wet weight) in fat of hatching-year (< 1 yr old) and adult (> 1.5 yr old) male common goldeneye collected on the upper Niagara River, New York. Data were taken from Foley and Batcheller (1988).

	Nov-Dec 1984				Feb-Mar 1985	
	sub-adult		adult		adult	
	N	mean	N	mean	N	mean
PCB	27	2.90	26	19.80	24	34.60
DDE	27	0.48	26	2.63	24	4.60
DDD + DDT	22	0.03	5	0.01	20	0.23
dieldrin	27	0.10	26	0.24	24	0.69
oxychlordane	27	0.04	26	0.22	24	0.25
mirex	1	0.01	21	0.08	14	0.11
heptachlor epoxide	27	0.04	25	0.11	24	0.23
HCB	25	0.04	26	0.09	23	0.15

sampled at the beginning and end of the over-wintering period. Violations of these assumptions would affect the validity of the statistical comparisons.

The implications of the common goldeneye data with respect to potential human exposure is not clear. The analytical data relate to mesenteric and subcutaneous fat deposits of the neck, breast, leg, subalar and ischio-pubic regions of individual birds (Foley and Batcheller, 1988). The NYSDH issues consumption advisories for waterfowl based on a 3.0 ppm PCB action level in poultry (fat) (Foley and Batcheller, 1988). There are no regular mammal or waterfowl contaminant monitoring programs in New York or Ontario.

5.1.6 CONTAMINANT SOURCES

The production and use of mirex was banned in the mid-1970s. Mirex levels in fish have declined gradually since the mid-1970s, but remain above the trigger level of 0.1 ppm for consumption restrictions (see OME and OMNR, 1991). Biomonitoring studies using clams indicate that the primary source of mirex is a sewer from an Occidental Chemical Corporation plant in Niagara Falls, New York (OME, 1988b). Variations in sediment concentrations and local hot-spots are consistent with current patterns in Lake Ontario.

Similarly the production and use of PCBs has been restricted since the 1970s, but fish data reviewed here indicate PCBs remain a widespread concern. PCBs are still in limited use and are found in STP effluents and landfill leachates (Government of Canada, 1991a,b). PCB levels were high in the tissues of clams at the outfall of Occidental's sewer (180-414 ppb) in Niagara Falls, New York, and Gill Creek (813-1120 ppb), downstream of the du Pont and Olin Corporation plants in Niagara Falls, New York (OME, 1988b). These are not the only sources of PCBs but they may be among the largest known sources of PCBs to the Niagara River and Lake Ontario.

Dioxins, including 2,3,7,8-TCDD, have been found in clams exposed in cages near the Pettit Flume sewer in North Tonawanda, New York (OME, 1988b). In 1987 2,3,7,8-TCDD levels in clams were 0.2 ppb, which represents an increase in concentration of an order of magnitude from 1985 (0.02 ppb). During the same period sediment levels increased from 3.6 ppb in 1985 to 9 ppb in 1987 (OME, 1988b). Advisories that the consumption of lake trout and brown trout at the Credit River and Jordan Harbour should be restricted due to 2,3,7,8-TCDD concentrations were issued for the first time in 1991. Since both sites are

in the western end of Lake Ontario, these findings appear to be consistent with the suggestion that the currents and predominant water circulation patterns will have a large role in the distribution of dioxins and accumulations in Lake Ontario fish (Jaffe and Hites, 1986).

Mercury appears to be a nearshore problem in the eastern basin and many of the industrialized harbours in the western basin. Whether mercury is derived from natural sources or the result of human activities is not clear. Probable human sources include industrial operations in western areas and sediments contaminated by former industrial operations and STPs in the Bay of Quinte watershed (Bay of Quinte RAP Team, 1990).

Rasmussen et al. (1990) demonstrated that much of the between-lake variation in PCB levels in lake trout tissue results from differences in the length of the pelagic food chain. Lake trout have higher PCB levels when alewife, smelt or other planktivorous fishes are present along with the zooplankter *Mysis relicta* (e.g., Lake Ontario), than in *Mysis* free lakes. These findings are consistent with the view that levels of contaminants such as PCBs in aquatic organisms reflect biomagnification from each trophic level, in this case by a factor of 3.5 times for each trophic level. If these findings, which were derived from comparative studies across a series of lakes, can be applied to a single lake through time, then they imply that lake trout contamination levels will decrease little until serious efforts are made to reduce the level of planktivory in Lake Ontario.

Large scale manipulations of the food-web have been contemplated or undertaken for a variety of reasons in the Great Lakes. One common rationale is to the control populations of exotic (non-native) species such as alewife and smelt by stocking large numbers of piscivorous species. For example, the introduction of Pacific salmon (coho and chinook)

species, which are also exotics, was initially justified in part as a means of controlling exploding alewife populations in the Great Lakes. Thus the idea of manipulating food-webs to reduce the body burdens of contaminants in lake trout is likely to be taken seriously.

5.1.7 CONCLUSIONS

Restrictions on fish consumption in Lake Ontario are caused by mirex, PCBs, mercury, and dioxin, specifically the 2,3,7,8-TCDD isomer. Mirex is currently responsible for approximately 40% of the restrictions and mercury is responsible for about 35% of the consumption restrictions reported in Lake Ontario fish. Restrictions due to organic contaminants predominate amongst the offshore salmonines in the main basin of the lake whereas mercury restrictions are prominent among the nearshore species in the Kingston Basin and the Bay of Quinte.

The database is sufficiently large and detailed in Ontario, in terms of species tested, locations, and contaminants tested, to established with confidence the causal pollutants (as defined in this study) restricting fish consumption in Lake Ontario. The data available from New York were less suitable in terms of locations tested, species, and specific contaminants.

5.2 TAINTING OF FISH AND WILDLIFE FLAVOUR

5.2.1 DEFINITION AND DETERMINANTS OF TAINTING

According to recently published International Joint Commission (IJC) listing/delisting criteria (IJC, 1991), the use impairment "tainting of fish and wildlife flavour" is defined in the following manner.

Listing Guideline

When ambient water quality standards, objectives, or guidelines, for the anthropogenic substance(s) known to cause tainting, are being exceeded or survey results have identified tainting of fish or wildlife flavour.

Delisting Guideline

When survey results confirm no tainting of fish or wildlife flavour.

5.2.2 ESTABLISHED GUIDELINES

In its discussion of tainting substances, the Great Lakes Water Quality Agreement (GLWQA, 1989) refers to maintenance of levels of phenolic compounds below 1.0 mg/L in public water supplies. This is to protect against unpleasant taste and odor of domestic water supplies. Substances which enter lake water as a result of human activity and have tainting effects on edible aquatic organisms

> "should not be present in concentrations which will lower the acceptability of these organisms as determined by organoleptic tests." (GLWQA, 1989).

However, no specific limits have been established nor have standardized methods or organoleptic testing procedures been developed. Interestingly, the Canadian Department of Fisheries and Oceans employs gas chromatographic analysis to assess organoleptic quality of commercial fish and fish products.

New York State has six water quality standards for C class waters (suitable for fish propagation) to protect against tainting of fish flavour. These are outlined in Table 5.2.1.

Table 5.2.1. New York State fishing tainting standards.

Substance	Limit (μg/L)
Aminocresols	5
Chlorinated benzenes	50
Total unchlorinated phenols	5
Total chlorinated phenols	1
Pentachlorophenol	0.4

Source: New York Department of Environmental Conservation (1987).

5.2.3 SURVEY OF IMPAIRMENT

The Great Lakes Water Quality Board (GLWQB) has recently provided a summary of tainting of fish and wildlife flavour in selected areas of concern in the Lake Ontario region. As summarized in Table 5.2.2, only Buffalo River fish are likely impaired.

Table 5.2.2 Summary of tainting of fish and wildlife in Lake Ontario region.

Location	Status
Eighteen Mile Creek	• currently being evaluated through public participation program
Oswego River	• currently being evaluated through public participation program
Rochester Embayment	• currently being evaluated through public participation program
Buffalo River	• likely impaired
Hamilton Harbour	• no reports of tainting
Toronto Harbour	• no reports of tainting
Port Hope	• no reports of tainting
Bay of Quinte	• no reports of tainting

Source: GLWQB (1989a).

5.2.4 FISH SPECIES

An evaluation of fish tainting was reported in one study (Jardine and Bowman, 1990). Although not performed with Lake Ontario fish, its results demonstrate the difficulty of such an evaluation. As a component of the Spanish River Remedial Action Plan, the study assessed tainting using a sensory evaluation method known as the triangle test. Three samples were presented to a panelist, two the same, and one different. The sample categorized as "different" is then judged as being "control" or "exposed". The study found that the sensory panel could not consistently distinguish between the exposed and control fish and could not distinguish a noticeable taint in those tests where they may have correctly differentiated between the control and exposed samples (Jardine and Bowman, 1990). Difficulties with such organoleptic problems are discussed further in Section 5.9.

5.2.5 WATERFOWL SPECIES

Contaminant levels in waterfowl species are dependent on food sources for a particular species. Since established taint limits apply to fat tissue of chickens, it is difficult to set human tolerance levels for waterfowl (GLWQB, 1989b). Nevertheless, it is noted that lipid-based contaminant concentrations in waterfowl greatly exceed limits prescribed for poultry.

5.2.6 CONCLUSIONS

As defined, the tainting of fish and wildlife flavour impairment is difficult to assess or list. An explanation for the absence of reports on this impairment is the preventive objective under Annex 1 of GLWQA (1989) to keep levels of phenolic compounds below 1 μg/L.

5.3 DEGRADED FISH AND WILDLIFE POPULATIONS

5.3.1 DEFINITIONS

The International Joint Commission's (IJC) guideline for use impairment 3, "degraded fish and wildlife populations", are shown below.

Listing Guideline

When fish and wildlife management programs have indentified degraded fish or wildlife populations due to a cause within the watershed. In addition, this use will be considered impaired when relevant, field-validated, fish or wildlife bioassays with appropriate quality assurance/quality controls confirm significant toxicity from water column or sediment contaminants.

Delisting Guideline

When environmental conditions support healthy, self-sustaining communities of desired fish and wildlife at predetermined levels of abundance that would be expected from the amount and quality of suitable physical, chemical and biological habitat present. An effort must be made to ensure that fish and wildlife objectives for Areas of Concern are consistent with Great Lakes ecosystem objectives and Great Lakes Fishery Commission fish community goals. Further, in the absence of community structure data, this use will be considered restored when fish and wildlife bioassays confirm no significant toxicity from water column or sediment contaminants.

Fish and wildlife populations can be impaired due to causal factors such as habitat loss or disruption, poor water quality, altered food chains, etc., which often act indirectly on the target organisms through other trophic levels. Bioassays indicating that contaminants in the sediments or water are directly toxic to fish and wildlife populations is a second means of listing this use as impaired.

The guidelines for recommending listing/delisting of Areas of Concern (AOCs) based on degraded fish and wildlife populations are formulated to emphasize fish and wildlife management goals as stated by the relevant agencies and to be consistent with the goals of the Great Lakes Water Quality Agreement (GLWQA) and the Great Lakes Fishery Commission. Although the bioassay criterion recognizes that toxic substances have a disintegrative effect physiologically and ecologically and that it is important that they be prohibited, it is a narrow view of the meaning of the term degraded. Every effort should be made to obtain community structure data, particularly with respect to delisting this use impairment.

Fish and wildlife management goals are generally based on some baseline conditions during a period when few stresses were operative. Hence it is important to note that population assessments are made relative to these baseline conditions. If conditions prior to European settlement in the Lake Ontario basin are taken as the relevant baseline conditions, then every locale in Lake Ontario now has degraded fish and wildlife populations. Some species native to the lake have been rendered extinct within it.

5.3.1 DEGRADED FISH POPULATIONS

Changes in the structure and abundance of the offshore (Christie, 1972, 1973, 1974; Christie et al., 1987; O'Gorman et al., 1987) and nearshore (Hurley and Christie, 1977; Whillans, 1979; Holmes, 1988) fish communities in Lake Ontario have been well documented. These reviews will be summarized below in order to highlight the direction of changes and their causes. In most cases estimates of relative abundance (e.g., catch-per-unit-

of-effort, or what was caught), which assume changes in catches at index stations reflect changes in population abundance, have been used as an indicator of population trends. Direct estimates of lake trout (Schneider et al., 1990) and alewife and rainbow smelt (O'Gorman et al., 1987) abundances, representing what is out in the lake, have been available for the last decade. Although contaminant effects on fisheries have been an issue for some time, they have not taken precedence over other human impacts on the fish community in Lake Ontario. However, there is concern that the absence of natural reproduction in stocked lake trout and reduced reproductive success in stocked coho and chinook salmon may be related to maternally derived contaminant burdens in eggs and fry. Investigations of this hypothesis will be discussed below.

Prior to European settlement in the late 1700s, the deep, cold, oligotrophic open waters of Lake Ontario were dominated by large piscivorous species such as lake trout, burbot and Atlantic salmon (Christie, 1973, 1974). The coregonines were also well represented as large planktivore and macrobenthos feeders in the offshore fish community. The nearshore community was more diverse and was dominated by basses, sunfishes, and pikes in marshy riverine and embayment habitats and nearshore reefs protected from the open waters (e.g., Holmes, 1988). The waters in these areas were more fertile (mesotrophic) than the offshore waters due to nutrient and materials recycling by natural phenomena. Many cyprinid species (small minnows) and coolwater species such as walleye and yellow perch made extensive use of protected nearshore habitats for feeding and reproduction. Most of the offshore residents homed to specific nearshore features such as reefs along the open shoreline and riffles in streams, for spawning.

Current fish communities are degraded relative to the above conditions. Lake trout, lake whitefish, and the cisco complex (*Coregonus* spp.) were important components of the commercial fishery that became excessively intense a century ago. This combined with nearshore habitat destruction and increased effects from sea lamprey predation led to the virtual extirpation of lake trout and burbot by the 1950s (Christie, 1972, 1973). Whitefish and cisco populations also collapsed but were not extirpated completely, although some of the deepwater ciscoes were extinguished. Atlantic salmon were extirpated by the 1870s, principally as a result of habitat destruction in spawning streams. As a result, the open waters were devoid of large piscivores until the early 1970s when stockings of salmonines began (Pacific salmon, lake trout, brown trout, rainbow trout). In the interim, the abundance of small-bodied, pelagic, exotic species such as alewife and rainbow smelt, exploded in offshore waters. The subsequent introduction and stocking of coho and chinook salmon was partly justified as a means of controlling alewife and smelt populations. Currently, coho and chinook salmon, lake trout, and alewife are the dominant species in the offshore fish community. Lake trout, chinook, and coho salmon populations are maintained by annual stockings which approach a combined goal of 8.2 million ($\pm$ 5%) fish set for Lake Ontario (LeTendre and Savoie, 1990). Although some self-sustaining populations of rainbow trout and brown trout have been established in Lake Ontario, supplemental stocking is undertaken to maintain stocks throughout the lake. Atlantic salmon have been stocked recently by New York and Ontario in an attempt to begin the rehabilitation of this species in Lake Ontario.

The nearshore fish community has also undergone changes but not to the extent of the offshore community. Pike and bass were the primary species sought by the early sport fishery. Excessive harvest of these species combined with the destruction of marsh habitat led to drastic declines in the abundance of these valued species. Excessive eutrophication led to plankton blooms, which shaded out rooted aquatic plants, interfered with the vision of sight-feeding piscivores such as pike, and as a result of decomposition caused localized anoxia in the bottom waters of shallow areas such as the Bay of Quinte (Hurley and Christie, 1977). However, few nearshore fishes were extirpated because they were not as reliant on specific habitat features as were the offshore species. Species such as carp and white perch, which are relatively tolerant of deteriorating water and habitat quality, subsequently invaded and became abundant in areas such as the Bay of Quinte and Hamilton Harbour. The abundance of white perch in the Bay of Quinte has declined in recent years as a result of ecosystem changes initiated by reductions in point-source phosphorus loadings (Hurley, 1986).

The degradation of nearshore and offshore fish populations in Lake Ontario likely reached its most extreme point in the 1950s and 1960s. During the last two decades sea lamprey control was instituted, commercial fishing effort and harvest were reduced, a large salmonid stocking program was instituted, and inputs of nutrients and toxic contaminants were significantly reduced. Although the full effects of these programs are not clear yet, recent changes in community structure and abundance are in the direction of restoring earlier system status (Christie et al., 1987). For example, during the past 10 years there have been substantial increases in piscivorous populations of walleye and lake trout (Christie et al.,

1987). The increases in lake trout and Pacific salmons are the result of massive annual stockings and have contributed to a reduction in the size and abundance of rainbow smelt. Alewife abundance has apparently not yet decreased greatly from salmonine predation (O'Gorman et al., 1987).

Of the six of AOCs on Lake Ontario for which Stage I Remedial Action Plan (RAP) reports are available, five list degradation of fish populations as an impaired use (Niagara River (Ontario), Hamilton, Toronto, Bay of Quinte, Oswego) despite evidence that restoration of the offshore waters is occuring. Only the Port Hope RAP report indicated that this use was not impaired (Port Hope RAP Team, 1989). However, the Port Hope assessment was made relative to fish populations in other small harbours which are probably degraded; at best it indicates Port Hope is no worse than other degraded locales. No corroborating documentation was provided supporting the assessment that this use was not impaired in Port Hope. Stage I reports have not been submitted yet from the Rochester, Eighteenmile Creek, and Niagara River (New York) AOCs.

Current fisheries management activities in Lake Ontario include (1) population management, (2) sea lamprey control, and (3) habitat management (see New York State Department of Environmental Conservation (NYDEC), 1991 and Ontario Ministry of Natural Resources (OMNR), 1991). Population management activities consist primarily of stocking of salmonines, regulating minimum sizes of fish caught, biomass harvest, and exploitation rates (commercial and sport), and the assessment of population abundance. Sea lamprey control involves the chemical treatment of natal streams to kill the larvae and to assess larval abundances. Under habitat management are included protection of existing habitats and

rehabilitation and restoration of degraded habitats. Many of these activities are undertaken jointly by the OMNR, the NYDEC, the U.S. Fish and Wildlife Service, and the Canada Department of Fisheries and Oceans, although the former two agencies exercise primary management authority on Lake Ontario. Although these agencies are aware of the contaminants issue through fish consumption advisories (see Section 5.1), contaminants have not been considered causal agents in the degradation of native fish populations because most of the valued species were declining prior to the period of most intensive contamination, and because the relative impact of toxic substances on the fish populations is inferred to be much less than that of other adverse stresses such as habitat losses, over-exploitation and introductions of exotic species.

5.3.1.1 Contaminant Effects on the Reproduction of Stocked Salmonines in Lake Ontario. Significant losses due to embryo mortality and mortality at swim-up have been observed in fish culture facilities in most salmonine species since the mid-1960s (Skea et al., 1985). These mortalites are not related to culture practices, water quality, disease agents, or pathogens and have only been observed in hatchery or laboratory environments; they have not been reported in field studies of wild or feral populations. The repeated findings of some level of developmental mortality suggests a common, widespread etiology and chemical contaminants have been inferred as a causal agent (Mac et al., 1985; Skea et al., 1985).

The hatchability of salmonine eggs and fry survival in hatchery situations are thought to be controlled by the occurrence and magnitude of three syndromes: (1) blue-sac disease in sac fry; (2) egg hatch mortality; and (3) fry mortality at swim-up (swim-up syndrome).

Observations of swim-up mortality syndrome have been reported in lake trout, steelhead trout, and coho salmon from Lake Ontario (Frimeth, 1990; Mac, 1990; Marcquenski, 1990). Blue-sac disease has been observed in lake trout (Symula et al., 1990) and has been produced in lake trout fry exposed as eggs to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (Walker et al. 1991). Pre-hatching embryo mortality has been reported in cultured lake trout, steelhead trout, coho salmon, and brown trout from Lake Ontario (Frimeth, 1990; Simonin et al., 1990).

Investigations of contaminant-caused reproductive impairments in Lake Ontario salmonines were initiated in Ontario because of mortalities ranging between 20 and 30% at swim-up and premature hatching in coho salmon (Frimeth, 1990). The initial investigations used eggs collected in the fall of 1990 from 28 female Credit River chinook and coho salmon, which were fertilized and reared in separate lots from individual females. Swim-up mortality exceeded 25% in fry from 9 of 28 female chinook salmon and averaged 22% overall (112/500 hatched fry; Smith et al., 1991). Fry from 17 female coho salmon exhibited > 30% mortality, with an average of 46% for all 28 females (139/300 hatched fry; Smith et al., 1991). The pattern of mortality appears to be female specific, but contaminant analysis of the muscle, eggs, and livers of females and their fry have not yet been completed. A similar female-dependent mortality pattern occurring in Lake Michigan lake trout was apparently related to variation in maternal contaminant body burdens (Mac and Edsall, 1991).

Between 1975 and 1984 the NYDEC conducted studies to evaluate the reproductive potential and health of salmonines from Lake Ontario in relation to their relatively high contaminant burdens (Simonin et al., 1990). These studies used eggs and fry of brown trout, lake trout, steelhead trout (rainbow trout that do not have a stream resident juvenile phase in

their life cycle), chinook, and coho salmon and have focussed on PCBs, mirex, HCB, and other organochlorine contaminants commonly found in Lake Ontario fish. Egg mortalities in individual lots of brown trout ranged from 40 to 100% and as a result the NYDEC no longer relies on Lake Ontario brown trout as an egg source for its hatchery system. Coho salmon exhibited 50% or greater overall egg mortality. Lake trout experienced high mortalities as eggs and sac fry, with a very high incidence of blue-sac mortality and spring-spawned steelhead trout experienced swim-up mortality of 25-35%. These losses were generally female-dependent, but no simple correlations were found with concentrations of DDT, DDE, PCBs, dieldrin, or mirex in the eggs. The absence of correlations demonstrates the difficulty of establishing causality between contaminants and reproductive impairments in feral fish and highlights the utility of applying epidemiologic criteria to establish a cause-effect relationship.

Blue-sac disease is characterized by edema and hemorrhages of the yolk sac that are specific to the sac fry stage of lake trout. The etiology is poorly understood, however, Symula et al. (1990) isolated a pathologic myxobacterium, *Cytophaga* sp., from affected sac fry in New York. This organism may have been transmitted to the fry through the egg and/or sperm. Incubating eggs at 1.7-3.0 °C rather than 7-9 °C significantly reduced the incidence of blue-sac disease and associated mortality and seems to rule out contaminants as a causal factor. However, Walker et al. (1991) clearly demonstrated that 2,3,7,8-TCDD causes lesions in sac fry that are morphologically identical to blue-sac disease.

The three mortality syndromes identified in the eggs and fry of Great Lakes salmonines are thought to be related to contaminants. These syndromes appear to be female-dependent within the same stock and egg dependent among lots from individual females. Taking into account differences in age, size, and lipid content, there is considerable variation in the tissue concentrations of chemicals among individual females exposed to the same concentration of chemicals. The earliest life stages, eggs and sac fry, seem to be most sensitive to the adverse effects of contaminants, which is consistent with studies of birds, mammals, and invertebrates (Government of Canada, 1991a,b).

5.3.1.2 Environmental Contaminants and Lake Trout Reproductive Failures. Restoration of a naturally reproducing lake trout population is the goal of a major international rehabilitation effort in Lake Ontario (Schneider et al., 1983). Lake trout have been stocked since 1973 and adults have been successfully reestablished but few of their naturally produced young survive (Marsden et al., 1988). Among Great Lakes salmonines, those from Lake Ontario, particularly lake trout, have the highest body burdens of organochlorine contaminants (Baumann and Whittle, 1988; De Vault et al., 1989; Niimi and Oliver, 1989; Whittle and Keir, 1991). Degraded interstitial water quality on spawning shoals or predation by alewifes on newly hatched fry (Sly 1988; Schneider et al. 1990) are possible explanations and are being investigated because lake trout have very specific spawning habitat requirements. However, there also is concern that contaminant effects on reproduction may be compromising lake trout rehabilitation efforts in Lake Ontario.

Mac and Edsall (1991) applied epidemiological criteria to infer the cause-effect relationship between organic contaminants and reproductive failure in lake trout from southeastern Lake Michigan. The criteria that they used - time order, strength of association, specificity, consistency on replication, and coherence - are consistent with our criteria for assessing the weight-of-evidence and inferring causality (see Chapter 4). Two distinct endpoints were used in their examination of lake trout reproductive failure: (1) the hatchability of eggs; and (2) fry survival. They concluded that maternally-derived PCBs, particularly specific congeners such as 3,3'4,4'-TCB, were causing significant reductions in the hatchability of lake trout eggs. Further, that organic contaminants with a mode of action similar to DDT, and of maternal origin, were causing significant swim-up mortality in fry. The findings of Mac and Edsall (1991) are summarized below because they provide the most convincing case of the role of contaminants in lake trout reproductive failure.

The hatchability of Lake Michigan lake trout eggs and fry survival were measured in the laboratory between 1975 and 1988 using eggs from wild females. These data show that hatchability improved from 60-70% in 1975 to almost 100% by 1988. The pattern of fry survival differed in that swim-up mortality first occurred in 1978 and remained high through 1981, resulting in survival rates of less than 20%. Swim-up mortality was sharply reduced after 1981 and survival rates rose to 70-100%. Concurrent measurements of contaminant residues in fish revealed a decreasing trend in DDT, total PCBs, and oxychlordane during the 1975-1982 period. Changes in egg hatchability support the time-order criterion, but the fry survival evidence is indeterminate.

The strength of association criterion relates to the degree to which the proposed cause and effect coincide in their distribution and takes into account the size of the effect observed in exposed versus non-exposed populations (Mac and Edsall, 1991). A comparative study of egg hatchability and fry survival among lake trout populations in the Great Lakes showed that egg hatchability was poorest in eggs from Lake Michigan and highest in eggs from Lake Superior. Fry survival was also poor in fry of Lake Michigan origin (averaging 4%) compared to fry from Lake Superior (59%). Egg hatchability and fry survival were consistent with the pattern of organochlorine concentrations, particularly PCBs, in both eggs and adult fish from the Lakes compared. There is also limited information from a laboratory study in which the hatchability of eggs (from four females) was inversely correlated with the concentration of 3,3',4,4'-TCB, a PCB congener, in the eggs.

The precision with which one variable produces a given outcome defines the specificity criterion used by Mac and Edsall (1991). A combination of laboratory and field studies provide evidence that PCBs can adversely affect hatching success in eggs of exposed adult fish. Furthermore, the 3,3',4,4'-TCB congener is known to cause embryonic mortality in birds and EROD induction in fish. (Ethoxyresorufin-o-deethylase (EROD) is a mixed-function oxidase enzyme that is sometimes used as a biomarker of organism exposure to toxic chemicals.) DDT and its metabolites induced mortality which closely resembles the swim-up mortality syndrome observed in lake trout.

Mac and Edsall (1991) also described evidence supporting the replication criterion, particularly as it applies to fry survival. Hatching rates in Lake Michigan lake trout were lower than normal (> 90% is considered normal) prior to 1988, indicating replication of

embryonic mortality over time. Swim-up syndrome was observed in every year between 1975 and 1987 except 1984 and 1986 and was observed in lake trout from other locations and other salmonines from similarly contaminated areas (e.g., lake trout fry from Sturgeon Bay, Lake Michigan, and Yorkshire Bar, Lake Ontario; chinook salmon from Lake Michigan).

The biological plausibility of the relationship or the coherence criterion, is supported by evidence from laboratory and field studies on lake trout and other classes of animals (Mac and Edsall, 1991). The relationship between egg and hatchability and maternally transfered contaminants was demonstrated in a laboratory study in which hatchability was significantly reduced (29% compared to 67% in the controls) by a high dose of organochlorine contaminants in the diet of lake trout from hatchery broodstock. The swim-up syndrome observed in lake trout is similar to reports in the mid-1960s of mortality induced by DDT. Furthermore, the species affected, trout and salmon, are large, have high lipid content, are piscivores, and, at least for lake trout, are long-lived. These characterisitics allow considerable accumulation of organochlorine contaminants, of which a substantial dose is passed on to the progeny through the large eggs of trout and salmon. The biological plausibility of the relation is also supported by numerous observations of reproductive dysfunctions in several classes of animals other than fish in relation to organochlorines such as PCBs.

Although contaminant levels have declined in Lake Ontario over the last 20 years (Baumann and Whittle, 1988; Government of Canada, 1991a), analyses for congeners of organochlorine contaminants have only recently begun so trends are not well documented. There are well known differences in the toxicity of specific congeners thus the observed

declines in contaminant levels do not mean that aquatic organisms are necessarily free of the adverse effects of those contaminants. The work of Mac and Edsall (1991) on the reproductive impairment of salmonines in Lake Michigan and the other Great Lakes provides a strong case for the role of coplanar PCBs, particularly the 3,3',4,4'-TCB congener, and unknown contaminants that may have a mode of action similar to that of DDT. However, the relevance of congener-specific analysis depends on the quantities of specific congeners found in the tissues and it subscribes to the cellular receptor model of chemical effects. There are other possible mechanisms that also should be investigated.

5.3.2 DEGRADATION OF WILDLIFE POPULATIONS

Temporal trends in population abundance are available for two groups of wildlife: colonial fish-eating waterbirds and mink. Breeding populations of colonial waterbirds have been censused using ground and aerial counts. These counts are not conducted annually because they are labour and/or resource intensive. The relative abundance of mink populations can only be estimated using harvest statistics from commercial trappers. These data are generally confounded by several factors including trapline size, species trapability, trapping effort, fur prices, changes in habitat, and trapping methods (Government of Canada, 1991b; Wren, 1991). However, considering the difficulties associated with data collection, these methods of estimating abundances of colonial waterbirds and mink are generally accepted as useful.

Although most of the AOCs listed degraded fish and wildlife populations as an impaired use, this assessment was usually made on the basis of degradation of fish populations. There is no documented observational or anecdotal evidence of wildlife population degradation in the Bay of Quinte or Port Hope AOCs (Bay of Quinte RAP Team, 1990; Port Hope RAP Team, 1989). In Toronto, Hamilton Harbour, and the Niagara River, there is good documentation of the abundance of colonial waterbird populations. In these AOCs populations of some species have exceeded objectives and reductions have been proposed (ring-billed gulls in Hamilton Harbour) or implemented (Canada geese in Toronto) (Hamilton Harbour RAP Team, 1989; Metro Toronto RAP Team, 1988). Where contaminant and abundance data are available, the connection between contaminant levels and population abundance is generally weak.

5.3.2.1 Mink. Permanent wetlands with ample shoreline and emergent vegetation are the preferred habitat of mink (Government of Canada, 1991b). Since the average home range is about 2 km in length (Wren, 1991), mink rely on locally available food sources. Whillans (1982) estimated a net loss of about 4,000 acres or 57% of the baseline area for 38 marshes along the Canadian shoreline of Lake Ontario west of the Bay of Quinte between 1789 and 1977-79. Losses were heaviest west of Toronto, generally exceeding 70%, and lightest east of Toronto where they ranged between 24 and 33%, depending on the county.

Overall, mink harvest in Ontario has decreased between 1920 and 1988 to the current 25,000-30,000 per year (Government of Canada, 1991b). The harvest began declining in the 1950s from a high of approximately 60,000-90,000 to a low of about 15,000 in the mid-

1970s, followed by a moderate recovery. The average number caught per trapper parallels the harvest, declining from 7.8 in 1950-51 to 1.6 in 1981-82, increasing slightly to 2.1 in 1987-88 (Wren, 1991). The harvest data do not have sufficient resolution to separate temporal trends in shoreline populations from inland populations, which are less likely to come into contact with contaminated food sources from Lake Ontario. Nevertheless, it is generally believed that the abundance of shoreline populations has declined (Colborn et al., 1990; Government of Canada, 1991b; Wren, 1991).

A study of mink and muskrat harvests between 1970-71 and 1984-85 compared two areas with high risk of PCB exposure to two areas with low risk PCB exposure (Jones, unpublished manuscript cited in Government of Canada, 1991b). Mink harvest in areas where the risk of PCB exposure was high was significantly lower ($P < 0.05$) than in areas where the risk was low. In contrast, the high risk areas did not necessarily have lower muskrat harvests, which implies that habitat changes were not a factor since mink and muskrat utilize similar habitat. However, muskrats are vegetarians and therefore do not depend on fish and other aquatic organisms as food sources. The areas of high risk included townships along the north shores of Lakes Erie and Ontario and the St. Lawrence River and the areas of low risk included adjacent townships not bordering the Great Lakes.

There is a paucity of residue burden data in wild mink and the data that are available are generally vague with respect to associated sampling details. PCB 1254/1260 in mink homogenates from Lake Ontario ranged from 0.08 to 1.02 μg/g, wet weight, and from 3.2 to 56.7 μg/g, on a lipid weight basis (D.V. Weseloh, unpublished data cited in Wren, 1991). Livers from five mink captured between Whitby and Cobourg contained 21 μg/g on a lipid

weight basis (Government of Canada, 1991b). Foley et al. (1988) reported that mean PCB 1254/1260 levels (wet weight basis) in the liver and adipose tissues of mink from New York trapped between 1982 and 1984 ranged from 0.4 to 95 μg/g and 0.03 to 7.9 μg/g respectively. The highest organ levels of PCBs (> 1.1 μg/g) were collected from habitats in Oswego (which borders Lake Ontario), Saratogo, and Orange Counties. PCB levels in mink were correlated with concentrations in fish; collection stations were spaced 10-20 km apart (Foley et al., 1988), i.e., several times the length of a home range (see above).

The concentrations of PCBs in wild mink populations collected from Lake Ontario habitats are generally sufficient to experimentally induce adverse effects on reproduction. Aulerich and Ringer (1977) reported that 5 μg/g of Aroclor 1254 in the diet of laboratory mink reduced reproduction when fed only 4 months and that 15 μg/g PCB totally inhibited reproduction and resulted in 31% adult mortality. These results are consistent with findings by Hornshaw et al. (1983) that 13.3 μg/g PCB was present in the fat of female mink exhibiting impaired reproduction and no kit survival in the laboratory.

Wren (1991) used the same epidemiological criteria as Mac and Edsall (1991) to assess the cause-effect linkage between chemicals and mink populations in the Great Lakes basin. He concluded that the results of toxicological experiments on mink reproduction and mortality strongly supported the specificity criterion, but that such effects have yet to be observed in the field because the available data (harvest records) have poor resolution. The coherence criterion was supported by toxicological studies as well as some field evidence of lower mink harvests in "high risk" populations. There was only weak evidence for the strength-of-association, consistency on replication, and time-order criteria. Overall, Wren

(1991) was not able to infer a clear cause-effect linkage between chemicals and wild mink populations, which he attributed to the absence of good field data. The reproductive effects of contaminants on mink are discussed more fully in Section 5.5.

5.3.2.2 Colonial Waterbirds. Double-crested cormorants were established at the eastern end of Lake Ontario by the 1940s-1950s when the population was estimated at 220 breeding pairs (Government of Canada, 1991b). Because of a complete reproductive failure between 1950 and 1977 the breeding population crashed to 22 pairs by 1974 (Price and Weseloh, 1986). The population had begun to recover by 1979 when 315 pairs were counted and by 1990 had exploded to 6,700 breeding pairs in seven active colonies (Government of Canada, 1991b). Since 1979, productivity has been within the range considered normal (1-2 young/pair) varying from 1.3 to 2.3 young/pair (Government of Canada, 1991b), which partly explains population growth since 1974 (Price and Weseloh, 1986). The expansion of the cormorant population has been accompanied by movement from the eastern islands (Pigeon, Little Galloo, Scotch Bonnet), through the central basin (Peter Rock) and into the western end of Lake Ontario (Hamilton Harbour). The number of breeding pairs now greatly exceeds the numbers estimated in the period when there was little known usage of organochlorine chemicals (1940s-1950s). Between 1976/77 and 1990 the average annual rate of increase in the cormorant population was 40.4% per year (Blokpoel and Tessier, 1991).

The data set on contaminants in cormorant eggs is not sufficient to establish temporal trends in organochlorine residue levels. Only four sites (three in eastern Lake Ontario) were sampled once each between 1972 and 1989. Comparison of the data from Scotch Bonnet

Island (1972) and Pigeon Island (1981) suggests that PCB levels did not change (17.8-18.3 ppm, wet weight) and that DDE levels decreased from 9.4 to 3.7 ppm, wet weight (Government of Canada, 1991a). Most contaminant levels in cormorants collected from Hamilton Harbour in 1989 were lower than at the other sites.

The abundance of **black-crowned night-heron** populations in Lake Ontario has only been determined for two colonies. Between the late 1960s and the early 1970s the number of nesting pairs declined along the Toronto waterfront, but then increased steadily through the 1980s to over 600 pairs in 1988 (Burgess, unpublished data cited in Government of Canada, 1991b). A second colony on Pigeon Island consisted of about 125 pairs between 1961 and 1963. Only 27 pairs remained by 1969 and they experienced complete reproductive failure (Government of Canada, 1991b). A modest recovery to 71 pairs occurred by 1977. Between 1972 and 1976 the reproductive success of the Pigeon Island colony was poor, in that more eggs were laid per pair (4.6-6.0) than average and the number of young fledged per active pair was low, ranging from 0.5 to 1.4 (I. Price, unpublished data cited in Government of Canada, 1991b). This can be compared to the Toronto colony in 1988 in which 4.0 eggs were laid per pair and 2.0 young fledged per pair (Burgess, unpublished data cited in Government of Canada, 1991b).

Contaminant data in night-heron eggs are fragmentary and are insufficient for establishing a relationship with population abundance in the Lake Ontario colonies. Annual mean DDE levels of eggs from Pigeon Island ranged from 4.5 to 12.4 ppm in the mid-1970s, dropping to 4.8 ppm by 1982 (Government of Canada, 1991b). A concommittant increase in eggshell thickness, from 14-16% thinner to only 6% thinner than prior to DDT use,

occurred as DDE levels declined. Gilbertson et al. (1976) reported a single abnormal chick with a crossbill among 72 night-heron chicks examined from the Pigeon Island colony in 1972 and 1973. Mean DDE (4.1-4.8), dieldrin (0.24-0.31), mirex (0.75-1.0), HCB (0.04-0.06), and PCB (18.9-24.4) concentrations (ppm, wet weight) varied little among eggs collected from colonies on Pigeon Island, Little Galloo Island, and the Niagara River in 1982.

The last successful nesting of **bald eagles** on Lake Ontario probably occurred around 1951 (Quilliam 1965, cited in Government of Canada, 1991b). Weekes (1974) surveyed the historical breeding range south of a line from Owen Sound to Pembroke, including the Bruce Peninsula, from 1969 to 1973 and reported finding only 10 active nests and 22 unsuccessful nests in an area esimated to have 100 active, generally successful nests prior to 1950. Bald eagles have not attempted to nest along the Lake Ontario shoreline in the 1980s despite the presence of apparently suitable habitat (Government of Canada, 1991b).

The Ecological Objectives Committee (1989) and Colborn (1991) applied epidemiologic criteria to evaluate the relationship between organochlorine chemicals and bald eagle populations in the Great Lakes. There was strong evidence that DDE caused eggshell thinning and reproductive impairments and that dieldrin resulted in excess adult mortality and that both together contributed to the decline of bald eagle populations. Metabolic changes, deformities, and behavioral changes in adults were also linked to organochlorine contaminants, notably DDT and its metabolites, based on evidence from areas in which eagles still reside (see Section 5.5; Government of Canada, 1991b; Colborn, 1991). However, the evidence with respect to the contribution of PCBs to reproductive problems is not clear and may require congener-specific tissue analysis. Egg contaminant data are not available for

Lake Ontario and only fragmentary data are available from elsewhere because as a threatened species eggs cannot be collected annually for contaminants analysis.

The bald eagle population on Lake Ontario is clearly degraded relative to historic conditions, although historic abundances are not known precisely. It has been proposed that the bald eagle be used as an integrative indicator of persistent toxic contamination in the Great Lakes (Ecological Objectives Committee, 1989). The reestablishment of populations in their former territories would be a useful indicator of improved conditions reflecting declines in organochlorine levels. The recommended objective for Lake Ontario is restoration such that a minimum of 20 bald eagle territories are occupied.

The productivity of **herring gull** colonies has been used as an index of abundance changes since 1972 because detailed estimates of abundance have not been made. Productivity exceeding 1.0-1.2 fledged young per pair is indicative of an expanding population, 0.8-1.0 young per pair is charactersitic of a normally reproducing, stable population, and less than 0.8 young per pair is indicative of a declining population (Government of Canada, 1991b). Between 1972 and 1975, reproductive success was low, ranging from 0.06 to 0.21 young per pair in colonies on Scotch Bonnet, Snake, Brother's, and Black Ant Islands, and Presqu'ile Park at the eastern end of Lake Ontario. Productivity rose above the 0.8 young per pair required for population stability in colonies on Mugg's, Snake, and Scotch Bonnet Islands in 1977 and since 1978 has varied between 0.86 and 2.13, indicating that all colonies are expanding (Government of Canada, 1991b). Abundance estimates are also consistent with these indices, showing an increase from 520 pairs in 10 colonies in 1976 to 1540 pairs in 15 colonies in 1987 (Blokpoel et al. in press, cited in

Government of Canada, 1991b). The average annual rate of increase in the herring gull population of Lake Ontario varied between 8.1 and 8.7% per year during the 1976/77-1990 period (Blokpoel and Tessier, 1991).

Herring gull eggs have been collected for contaminant analysis from six colonies on Lake Ontario and one in the Niagara River since 1974. Data from Snake Island and Mugg's Island show that in 1989 total PCB, DDE, mirex, HCB, and dieldrin concentrations were about 80% less than 1974 levels in eggs from both colonies (Government of Canada, 1991a). Most of the organochlorine contaminants declined sharply in the 1970s, followed by a small increase in 1981-82 and then a much slower decline or levelling off in the last 4-5 years. This temporal trend was consistently observed in eggs from all colonies in Lake Ontario and is consistent with patterns observed in colonies on the other Great Lakes (Government of Canada, 1991a). The reasons for the increase in 1981-82 are not clear. While contaminant levels are much lower relative to 1974, mirex and TCDD residues in eggs from Lake Ontario colonies are higher than those in eggs from other lakes by at least an order of magnitude (Government of Canada, 1991a).

The herring gull data provide circumstantial evidence on their own of a relationship between contaminant levels in tissues and population abundance. However, contaminant monitoring did not begin until 1974, which may have been well after peak contamination levels had occurred in herring gull tissues.

There are few estimates of **ring-billed gull** abundance in the Great Lakes. At the beginning of the 20th century they were rare (Government of Canada, 1991b). By 1976 the population on Lake Ontario was estimated to be 40,787 pairs in 4 colonies and by 1990 ring-

billed gulls were the most abundant species with 172,712 pairs in 15 colonies (Blokpoel and Tessier, 1991). The average annual rate of increased varied between 12.3 and 13.3% per year during this period (Blokpoel and Tessier, 1991).

Neither the reproductive success of ring-billed gulls nor egg contaminant residues have been monitored to the same extent as herring gulls. During the 1970s, Gilbertson et al. (1976) reported that about 27 chicks of more than 4,500 examined (ca. 0.6%) in the Mugg's Island colony had a leg deformity similar to perosis in poultry, which prevented them from standing or walking properly. The organochlorine residue levels in ring-billed gull eggs from two colonies were neither consistently higher nor lower than levels in herring gull eggs from the same locations (Table 5.3.1). However, there is insufficient information with respect to residues in ring-billed gull tissues to directly determine the role of contaminants, if any, in the abundance changes observed. Although the Mugg's Island and Hamilton Harbour colonies are close geographically, differences in contaminant levels may reflect spatial changes rather than temporal changes in contaminant loadings.

There were 47 breeding pairs of **Caspian terns** in two colonies on Lake Ontario in 1976-77, which underwent an increase to 765 pairs by 1990, an average annual rate of increase of 22.1% (Blokpoel and Tessier, 1991). Gilbertson et al. (1976) reported a single abnormal chick with a crossbill among about 100 examined on Pigeon Island in 1972. Although concentrations of DDE and PCBs in eggs from Pigeon Island declined significantly between 1972 and 1981 ($P \leq 0.05$), these eggs contained higher levels of PCBs (39.3 ppm, wet weight) and mirex (1.57 ppm, wet weight) than eggs from other lakes in 1980-81 (Struger and Weseloh, 1985, cited in Government of Canada, 1991a). The relationship

Table 5.3.1. Comparison of mean organochlorine levels ($\mu g/g$) in ring-billed gull (RBG) and herring gull (HG) eggs from two Lake Ontario colonies. Data were taken from Bishop et al. (1991).

	Mugg's Island 1979		Hamilton Harbour 1984	
	RBG	HG	RBG	HG
Sample size	24	10	14	1
Lipid content (%)	9.9	8.7	8.7	8.7
α-chlordane	0.067[a]	0.12[b]	0.061	0.02
oxy-chlordane	0.076	0.26[b]	0.095	0.18
HCB	0.12	0.20	0.044	0.065
DDD	0.11	0.12	0.031	0.02
DDE	3.02	9.04	3.2	5.12
DDT	0.026	0.079	0.032	0.02
dieldrin	0.52	0.221	0.40	0.2
heptachlor epoxide	0.08	0.062	0.10	0.04
β-hexachlorocyclohexane	0.028	0.046	0.004	0.02
mirex	0.74	1.8	0.50	1.34
PCB 1260	21.2	65.4	15.2	35.4
PCB 1254:1260	27.2	76.2	18.8	43.6

[a] Mean based on 10 samples.
[b] Mean based on 8 samples.

between organochlorine residues and productivity is not known since the reproductive success of the Lake Ontario colonies has not been studied.

The estimated population of **common terns** in the lower Great Lakes appears to have increased from the beginning of the 20th century to a peak of 16,000 breeding pairs in the early 1960s. Declines began in 1965 and by 1976 there were only 1,300 pairs on Lake Ontario, which decreased further to 940 pairs in 1987 (Courtney and Blokpoel, 1983, cited

in Government of Canada, 1991b). A small recovery to 1,150 pairs occurred by 1990 (Blokpoel and Tessier, 1991). Over the 1976 to 1990 period the common tern population decreased at a rate of 1.9-2.0% per year, in contrast to other colonial waterbird populations (Blokpoel and Tessier, 1991).

Data on reproductive success in five Lake Ontario colonies during the 1972-77 period indicates that productivity was less than 0.92 chicks fledged per breeding pair, which is lower than the 1.1 chicks per pair necessary to maintain a stable population (Government of Canada, 1991b). Only the Toronto Harbour colony, with a productivity of 1.18 chicks per breeding pair, could be considered a stable, normally reproducing population. The lack of reproductive success was due to egg failure and the disappearance of eggs and chicks. Gilbertson et al. (1976) reported finding 11 abnormal chicks among about 844 (ca. 1.3%) examined in six colonies between 1971 and 1973. Most of the deformities were crossed bills, but chicks with shortened mandibles, small eyes, and extra toes were also observed. The proportion and variety of abnormalities were the greatest of all species examined.

The role of organochlorine contaminants in the decline of common terns in Lake Ontario is not clear. Despite the early and ongoing concern about the status of common terns in Lake Ontario, eggs have not been collected annually from the same sites for contaminant analysis (Table 5.3.2) so temporal trends in contaminant levels cannot be established. The data that are available tend to suggest that most organochlorine residues have declined substantially (80-90%) over the period of concern, but these declines might also be related to a spatial rather than a temporal pattern of contaminant concentrations. The 1981 levels of PCBs, DDE, and mirex in tern eggs from Lake Ontario were significantly higher

Table 5.3.2. Comparison of mean organochlorine residue concentrations ($\mu g/g$, wet weight) in common tern eggs from Lake Ontario. Data were taken from Bishop et al. (1991).

Contaminant	Hamilton Harbour 1971	Mugg's Island 1972	Toronto Islands 1973	Presqu'ile Park 1975	Leslie St.[a] Spit 1981
Sample size	25	5	6	4	10
Lipid content (%)	7.4	7.0	5.9	7.6	NR
HCB	0.95	0.73	0.14	0.14	NR
DDD	0.58	0.24	0.17	0.054	NR
DDE	16.8	18.9	5.7	2.98	2.46
DDT	0.30	0.36	0.28	0.044	NR
dieldrin	0.54	0.40	0.34	0.5	0.14
heptachlor epoxide	0.098	0.24	0.058	0.062	NR
mercury	1.12	1.06	0.56	0.29	NR
PCB 1260	96.6	87.8	22.85	25.5	10.4

NR - not recorded.
[a] Geometric mean. All other data are arithmetic means.

($P < 0.05$) than in eggs from colonies on the other Great Lakes (Weseloh et al., 1989). But organochlorine residue levels in common tern eggs were lower than in eggs of Caspian terns or herring gulls from Lake Ontario. A combination of factors including loss of nesting habitat due to high water levels and encroachment by vegetation, human disturbance, and competition for nesting sites with the expanding ring-billed gull population may also be responsible for the decline (Morris et al., 1976; Government of Canada, 1991b).

Six bird species in Lake Ontario have experienced population declines in the 1960s and five had subsequent recoveries in the late-1970s and 1980s. These species include

double-crested cormorants, black-crowned night-herons, herring gulls, ring-billed gulls, and Caspian terns. The common tern population declined throughout the 1970s and 1980s, but recovered slightly in 1990. Bald eagles were extirpated from Lake Ontario by the 1950s. All of these birds are long-lived, fish-eating species.

Contaminant related reproductive impairments have been inferred based on the weight-of-evidence in colonies of double-crested cormorants, black-crowned night-herons, and bald eagles on Lake Ontario. The various reproductive impairments including eggshell thinning, chick and embryo mortality, growth retardation, congenital abnormalities, and aberrant adult behavior are discussed in Section 5.5. Through the late 1950s and 1960s biologically significant eggshell thinning associated with DDE was observed in double-crested cormorants, black-crowned night-herons, and herring gulls as well as common terns (Price and Weseloh, 1986; Government of Canada, 1991b). In the 1960s and 1970s herring gulls, black-crowned night-herons, and common terns also exhibited symptoms of developmental toxicity consistent with PCB effects (Government of Canada, 1991b). Population increases in the late 1970s and 1980s coincided with reductions in organochlorine levels in eggs and the return of eggshell thickness to near normal and fewer observations of developmental toxicity. For example, herring gull reproduction improved as egg concentrations of 2,3,7,8-TCDD and PCBs declined.

The increase in ring-billed gulls and double-crested cormorant populations is unprecedented and does not seem to be related to reduced contaminant loadings alone. Populations of these two species have average annual rates of change of 12.3-13.3 and 40.4% respectively compared to rates of 1.9 to 22.1% in the other species. An increase in the

abundance of small, pelagic fish such as alewife and rainbow smelt, which are major dietary items, over the same period (O'Gorman et al., 1987) may have contributed to the current success of these species (Government of Canada, 1991b).

The reproductive failures and population decreases of colonial waterbirds throughout the Great Lakes were investigated by Gilbertson et al. (1991). During the period of declining populations various lesions including ascites, increased pericardial fluid volume, subcutaneous edema, shorter tarsal length, liver enlargement, and necrosis were observed in embryos (Gilbertson, 1986). This suite of lesions is consistent with GLEMEDS or Great Lakes Embryo Mortality, Edema, and Deformities Syndrome (Gilbertson et al., 1991). The role of specific contaminants in GLEMEDS is not known, but non-ortho substituted PCB and dibenzo-*p*-dioxins and furans substituted at lateral positions are GLEMEDS-active compounds in herring gulls and other colonial waterbirds in the Great Lakes. Section 5.5 has a more complete discussion of contaminant related reproductive impairments including GLEMEDS.

5.3.3 CONCLUSIONS

Fish populations are degraded in terms of community structure and species abundances compared to conditions in the late 1700s. The most degraded point occurred about 30 years ago. Since that time various components of the ecosystem have improved to the point that stocked salmonines dominate the offshore coldwater community and elements of the native warmwater community (primarily pike and bass) have recovered some of their earlier status, although recent invaders such as white perch have partially usurped the role of these native species. Contaminants are not believed to be a major causal factor in the decline of fish

populations because the declines were well underway by the time contamination became intense. Habitat losses, over-exploitation, introductions of exotic species, eutrophication and other stresses are instead considered by fisheries experts to be the major reasons for fish community problems in Lake Ontario. Recent investigations are examining the relationship between PCBs and dioxins and the reproduction of stocked salmonines, particularly lake trout.

The evidence concerning the relationship between contaminants and the impairment of colonial waterbird and mink populations in the 1970s and 1980s is weak or circumstantial at best. There is a stronger case with respect to the reproductive effects of contaminants (as discussed in Section 5.5) from which population impairment has usually been inferred. Collectively the fact that the population numbers of various species apparently increased coincident with a decline of all or some of the organochlorine contaminants in their diets or body burdens may be taken as indicative of a relationship. The bald eagle provides the best example of a connection between contaminants (DDE/DDT) and population abundance since there was a simultaneous reversal of trends in contaminant levels and abundance. But the bald eagle example is derived from the other Great Lakes since eagles were extirpated around Lake Ontario well before chemical concentrations in tissues were measured. Furthermore, DDE/DDT concentrations have declined significantly since bald eagles were extirpated, but a concomittant recovery of the population in the Lake Ontario basin has not occurred even though there is suitable habitat. There does not appear to be a strong connection between any one contaminant and population abundance in any of the wildlife populations examined in this section, with the exception of DDE/DDT and bald eagles, and hence it is difficult to infer causality using population abundance data alone.

Degradation of fish and wildlife populations in Lake Ontario due to contaminants cannot be easily assessed with the available data. The fish data have sufficient temporal and spatial scope to be relatively confident in the abundance and community structure trends. But fisheries experts did not formally perceive contaminants as an issue until recently. This contrasts with the wildlife data because the monitoring programs were established due to concerns that contaminants were causing deformities and other reproductive problems in colonial waterbirds. These programs began in 1974, which may have been after the peak of contamination had occurred, and with one exception (herring gulls) contaminant burdens in eggs and population abundance have been monitored at irregular intervals. Furthermore, there is only anecdotal data relating to population sizes in the period before organochlorine use became intense (1940-1950), thus baseline conditions for colonial waterbirds are not as well known as they are for fish populations in Lake Ontario.

5.4 FISH TUMORS OR OTHER DEFORMITIES

5.4.1 DEFINITIONS

The International Joint Commission's listing/delisting guidelines for use impairment 4, "fish tumors or other deformities", are shown below.

Listing Guideline

When the incidence rates of fish tumors or other deformities exceed rates at unimpacted control sites or when survey data confirm the presence of neoplastic or preneoplastic liver tumors in bullheads or suckers.

Delisting Guideline

When the incidence rates of fish tumors and other deformities do not exceed rates at unimpacted control sites and when survey data confirm the absence of neoplastic or preneoplastic liver tumors in bullheads or suckers.

According to these guidelines a site may be listed as being impaired with respect to fish tumors or other deformities by satisfying either criterion, i.e., on the basis of a statistical comparison between exposed (impacted) sites and control (unimpacted) sites **or** on the basis of survey data indicating the presence of liver tumors. The background incidence rate of neoplastic or preneoplastic liver tumors is assumed, implicitly, to be zero. Delisting an area requires that both the statistical comparison **and** liver tumor survey criteria are satisfied and thus is more stringent than the listing guideline.

Terms such as lesion, neoplasm, tumor, and cancer are used throughout this section. The definitions below were taken from Duffus and Richardson (1990) and are listed from general to specific. Lesion and neoplasm are occasionally used synonymously. However, the former refers to a pathological disturbance resulting from an injury, an infection, or a

tumor and the latter is any new and morbid formation of tissue. A tumor is a growth of tissue forming an abnormal mass. Cells of a benign tumor will not spread and cause cancer whereas cells of a malignant tumor can spread within the organism and cause cancer. Cancer is a term used to describe more than 200 different disease processes which result from the development of a malignant tumor and its spread into surrounding tissues.

5.4.2 TUMOR INDENTIFICATION

Most of the tumor and deformity investigations of fish populations in Lake Ontario are conducted by research groups at the Canada Department of Fisheries and Oceans in Burlington, lead by V.W. Cairns, and at the Ontario Veterinary College in Guelph, which includes I.R. Smith (now at the Ontario Ministry of the Environment), M.A. Hayes, and H.W. Ferguson. Tumor diagnosis is based on gross pathology and histology using common criteria established at joint annual meetings of both groups (V.W. Cairns, Department of Fisheries and Oceans, Burlington, and I.R. Smith, Ontario Ministry of the Environment, Toronto, personal communications). The effectiveness of this collaborative harmonization is borne out by a comparison of incidence data from common sampling sites and times (Table 5.4.1).

Fish tumor data from the New York waters of Lake Ontario are sparse and come primarily from the Niagara River watershed. Tumors in New York are diagnosed using criteria established by the Registry of Tumors in Lower Animals (e.g., see Harshbarger and Clark, 1990) which differ subtly from those employed by Canadian authorities. Canadian diagnoses appear to be more conservative in assigning a cancerous designation to lesions.

Table 5.4.1. Comparison of tumor incidences reported for common sampling locations by the two Canadian research groups. ND means no data were available.

Species	Location	Incidence (%)	N	Year	Reference
		Epidermal papillomas			
white sucker	Grindstone Ck.	58[a]	225	1985	OVC - 1
white sucker	Grindstone Ck.	39[b]	168	1983-87	DFO - 2
brown bullhead	Hamilton Harbour	55	176	1985	OVC - 1
brown bullhead	Hamilton Harbour	48	225	1985	OVC - 3
		Liver tumors			
brown bullhead	Hamilton Harbour	7	ND	1985	OVC - 3
brown bullhead	Hamilton Harbour	2	124	1982-87	DFO - 1

References: 1 - Smith et al. (1989a); 2 - Government of Canada (1991b); 3 - Smith and Ferguson (1985).

[a] Incidence rate in sexually mature white suckers, i.e, approximately age 4 and older.
[b] Incidence rate in white suckers $\geq$ 7 years old.

OVC - Ontario Veterinary College research group.
DFO - Department of Fisheries and Oceans research group.

Tumor incidence rates in the same species from opposite sides of the border are not directly comparable. Tumor incidence rates generally are lower at Ontario sites than rates from comparable New York sites perhaps because of differences in the criteria used to diagnose tumors. Efforts to coordinate and apply common diagnostic criteria have been largely unsuccessful to date (I.R. Smith, personal communication).

The ubiquitous nature of low, perhaps undetectable, but biologically significant environmental contamination poses difficulty in defining appropriate control sites for fish tumor studies. As a result only two studies (Smith and Ferguson, 1985; Dickman and Steele, 1986) have statistically compared incidence rates at impacted locations to control sites. Many studies provide comparative data from reference areas on Lakes Erie, Huron, or Superior which were chosen because they are relatively unimpacted by environmental contaminants, i.e., there are no known industrial or municipal point sources or non-point sources nearby, at least with respect to the contaminants of concern. Both the DFO and OVC groups have recently found measurable tumor rates in fish populations from various reference sites used in the past (V.W. Cairns and I.R. Smith, personal commuications). Thus the assumption that there are locales with no neoplastic or preneoplastic liver tumors in the local fish with respect to the listing/delisting guidelines seems inappropriate.

5.4.3 TUMOR DEVELOPMENT IN FISH

The development of neoplasia in fish appears to be conceptually consistent with the 'resistant' model developed for mammalian studies (Hayes et al., 1987). This model has three stages in cancer development at which chemicals may have a role: initiation, promotion, and progression. Initiation begins with a mutation in the genetic material of the cell (DNA), possibly as a result of chemical exposure. If the mutation confers a growth advantage to the cell it will multiply to form a small focus, or group of mutant cells (Hayes et al., 1987). In the next stage, promotion, the mutant cells continue to proliferate forming large foci which can be called neoplasms. Promotion can be stimulated by chemicals such as PCBs or dietary

factors, but if the promotor disappears, then the cells can revert to normal and the neoplasm will regress (Hayes et al., 1987). Continued promotion and cell proliferation leads to the third stage, progression, in which additional genetic changes remove the mutant cells from host control over their growth. Once the progressive stage is reached, malignant cancers can form, invade adjacent tissues, and metastasize to distant tissues forming new cancerous cell colonies (Hayes et al., 1987). Although the parallels are not exact - for example, progression of neoplasms to malignancies is rare in fish (Hayes et al., 1987) - the similarities are sufficient to have prompted several investigations into the biochemical mechanisms for resistance in fish neoplasms (e.g. Black, 1983b; Maccubbin et al., 1988; Kirby et al., 1989a, b) and the behavior and effects on the host of skin and liver neoplasms (e.g. Smith and Zajdlik, 1987; Smith et al., 1989a; Hayes et al., 1990).

5.4.4 FISH TUMOR STUDIES IN LAKE ONTARIO

Documentation of fish tumors in Lake Ontario (Table 5.4.2) began in the 1970s with reports of thyroid enlargements (goiters) in coho salmon (Moccia et al., 1977) and epizootics of gonadal tumors in F_1 carp x goldfish hybrids (Sonstegard, 1977; Dickman and Steele, 1986; Down et al., 1988). Most of the work in the 1980s has focussed on the distribution and prevalence of epidermal papillomas and liver neoplasms in white suckers (Smith and Zajdlik, 1987; Kirby et al., 1988; Kirby et al., 1989a, b; Smith et al., 1989a, b; Hayes et al., 1990; Government of Canada, 1991b) and brown bullheads (Black, 1983; Maccubbin et al., 1988; Smith et al., 1989a; Government of Canada, 1991b). Liver tumors have been diagnosed in slimy sculpin recently (Fitzsimons, 1987). There is also a single report of

Table 5.4.2. Summary of fish tumor and deformity investigations in Lake Ontario, 1970-1990.

Fish Species	Type of tumor	Localization	Frequency range
white sucker	papilloma	skin, lip and body	0-64%, varies with age 10%, reference site at South Baymouth
	carcinoma	liver, bile ducts and liver cells	0-7.4% 0%, reference site at South Baymouth
brown bullhead	papilloma	skin	48-55% 14%, reference site at Long Point
	carcinoma	liver, bile ducts and liver cells	<1-20% 0%, reference site at Long Point
slimy sculpin	adenoma	liver	incidence not reported, off of Olcott, N.Y.
carp x goldfish	carcinoma	gonads	epizootics, about 100% in older F_1 males
coho salmon	hyperplasia	thyroid (goiter)	24-46%
rainbow trout	carcinoma	hepatocellular	3.1-8.1%, induced by sediment extract
lake trout	constrictions	testicles	63%

testicular constrictions in spawning lake trout from eastern Lake Ontario (Ruby and Cairns, 1983). Most of the epidermal and liver tumor incidence data in Sections 5.4.5.1 and 5.4.5.2 were taken from unpublished Department of Fisheries and Oceans manuscripts cited in

Government of Canada (1991b); these manuscripts were not available for critical appraisal (see Section 4.4 for description). The studies used in this section are summarized in Appendix E.

Tumors and deformities affecting skin, liver, and gonads have been indentified in seven fish species from 16 locations in western Lake Ontario and the Niagara River watershed (Table 5.4.2). Most of the information relates to nearshore locations because the species of interest is a nearshore inhabitant (brown bullhead) or because the species is more mobile but is aggregated at nearshore locations at certain points in its life cycle (spawning runs of white suckers and coho salmon). Gonadal constrictions in lake trout (Ruby and Cairns, 1983) and hepatocellular adenomas in slimy sculpin (e.g., Fitzsimons, 1987) are exceptions since both species are benthic offshore inhabitants and have a broad distribution in Lake Ontario. Epidermal papillomas are far more prevalent around Lake Ontario than liver or gonadal neoplasms. The prevalence of epidermal papillomas is roughly comparable in white sucker and brown bullhead populations, but the incidence of liver neoplasms appears to be lower in white suckers.

The relationship between environmental contaminants and neoplasms in wild fish populations (wild in the sense of self-sustaining native species rather than species maintained by stocking) throughout the Great lakes, including Lake Ontario, has been reviewed by Baumann (1984), Black (1984, 1988) and Harshbarger and Clark (1990). Several lines of evidence were supportive of the pollutant-carcinogen hypothesis including the temporal relationship between the discovery of tumors in fish and chemical production, experimental induction of carcinogenesis with pure compounds and contaminated sediment, the consistency

of the relationship between tumor prevalence and industrialization, and physiological experiments demonstrating that fish have the enzymes necessary to metabolize indirect-acting compounds into metabolites that are capable of inducing tumors. These authors concluded that epithelial tumors, particularly hepatocellular neoplasms, are correlated with exposure to chemicals from industrial and municipal sources, notably PAHs. Benthic species such as brown bullhead and white sucker, were most commonly affected and the risk of cancer increased with age, particularly in mature fish. Tumors were rarely reported in sexually immature fish from wild populations. Information about the effect of tumors on fish health is limited because estimates of mortality, reproductive impairments, and other ecological or physiological dysfunctions have not been made.

5.4.5 TUMORS IN LAKE ONTARIO FISH

The text in the following sections is subdivided on the basis of organ localization observed in Lake Ontario fish, e.g. skin, liver, gonads, and thyroid. When sufficient data are available they have been mapped so that the interrelationships between tumor type, fish species, geographic distribution, and chemical etiology can be understood.

5.4.5.1 Epidermal Papillomas. Three types of epidermal papillomas have been identified in white sucker and brown bullhead populations from Lake Ontario: (1) lip papillomas; (2) body papillomas; and (3) plaques. Lip papillomas are round or oval, markedly raised, firm and often smooth surfaced, focal lesions on or about the lips (Smith et al., 1989a). Body papillomas are focal, circular, distinct, firm, white, sharply demarcated,

discrete lesions, most commonly found on the dorsal lateral scaled body surfaces, in contrast to plaques which are pale, mucoid, low, variably shaped lesions found most commonly on unscaled body surfaces (Smith et al., 1989a). The incidence rate of lip papillomas (50-60%) is usually about twice that of plaques (20-30%) which in turn is greater than body papillomas (10-15%) when data for each category have been reported (e.g., Smith and Ferguson, 1985; Smith and Zajdlik, 1987; Smith et al., 1989a). Surveys of epidermal papillomas are much more extensive in white sucker populations than for brown bullhead populations (Figure 5.4.1). The data in Figure 5.4.1 are total epidermal papilloma incidence rates. At locations in which both species have been surveyed (Hamilton Harbour) comparable incidence rates are reported (35% in white suckers and 48% in brown bullheads). The frequency of occurrence is highest at locations in western Lake Ontario (e.g., 64% in Spencer Creek and 59% in Oakville Creek) and decreases moving eastward, with the exception of an anomalously high frequency (14.4%) in the Ganaraska River/Cobourg Creek area. Reference populations of white suckers at South Baymouth and brown bullheads at Long Point had papilloma frequencies of 10 and 14% respectively (Government of Canada, 1991b). Thus only east of Metropolitan Toronto does the prevalence of epidermal papillomas decline to or below background levels.

The white sucker data represent sexually mature fish only. Collections by the Department of Fisheries and Oceans (DFO) are for fish 7 years and older based on earlier observations of higher than expected incidences in larger fish (Government of Canada, 1991b). Approximately 150-200 fish were collected by the DFO at each site between 1983 and 1987. However, these collections appear to represent pooled samples from all collections

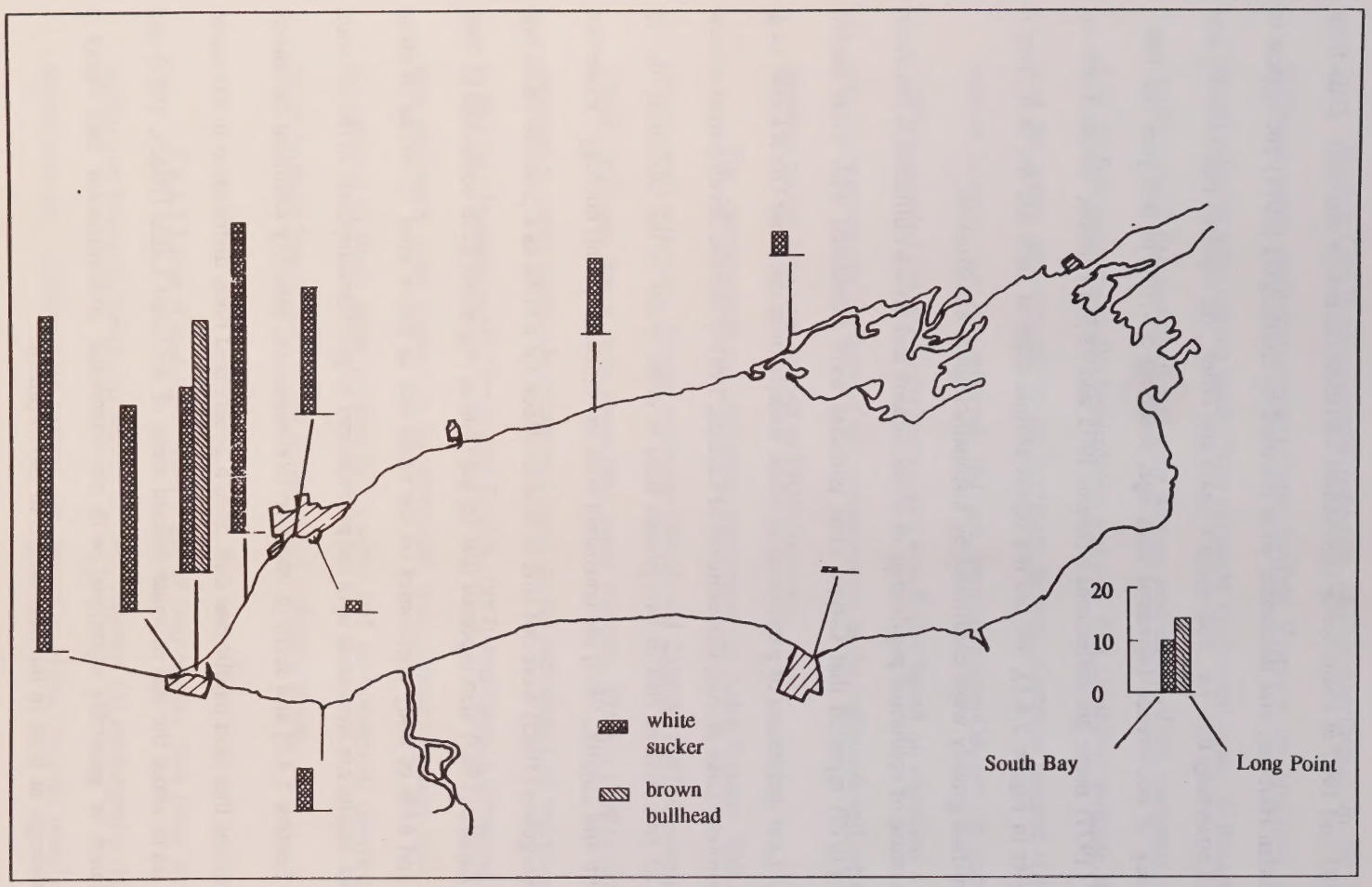

Figure 5.4.1. Prevalence of epidermal papillomas in wild white sucker and brown bullhead populations in Lake Ontario, 1982-1987.

between 1983 and 1987 at a site hence interannual variation cannot be assessed. Data from Toronto, Jordan Harbour, and Rochester were reported by Sonstegard (1977) and appear to consist of all spawning fish, i.e., fish about 4 years and older. He reported that both the size and frequency of occurrence increased with age, reaching 50-70% in 6-9 year old fish. Sonstegard (1977) made his collections between 1973 and 1976 (Toronto, Jordan Harbour, and Rochester in Figure 5.4.1), but did not report sample sizes at each site nor is it clear if tumors identified grossly were confirmed by histopathological examination.

The cause of epidermal papillomas in white suckers and brown bullheads is not clear. Sonstegard (1977) reported that C-type virus particles were associated with white sucker papillomas. Low incidences of papillomas in white suckers from the other Great Lakes (e.g. 1-10% in northern Lake Huron; Government of Canada, 1991b) and the development of new tumors on 23 and 60% of fish in two groups held in clean water in the laboratory for 12 weeks (Smith and Zajdlik, 1987) are consistent with an infectious (viral) etiology. However, Smith and Ferguson (1985) were not able to find evidence for a virus as a possible cause and Smith and Zajdlik (1987) also reported that lip papillomas regressed 22% compared to 79% of plaques and 64% of body papillomas on the same fish as above after 12 weeks in clean water. These results are consistent with the promotional stage of development in the resistant model (see Section 5.4.3) and seem to imply an environmental, possibly chemical causation. The observations that skin papillomas shift from a generalized body distribution to exclusive occurrence on or about the lips in industrialized areas of the Great Lakes (Black, 1988) and that prevalence is generally correlated with environmental contamination also imply a chemical etiology, at least in the later stages of development.

Black's (1983) experimental induction of skin papillomas on the heads of brown bullheads using sediment extracts from Buffalo River is the strongest support for the hypothesis that skin tumors result from exposure to contaminants. Although sample size is small (only 22 fish survived) and there was no replication of the treatment, after 18 months 5 of 22 fish were markedly hyperplastic and two fish exhibited multiple small papillomas over the entire area exposed to the sediment extract. Controls painted with the extraction solvent (sample size not reported) did not develop hyperplasticity or papillomas. Black (1983) suggested that the likely carcinogen was PAHs based on chemical analyses showing that PAHs accounted for most of the contamination in the sediments and other animal studies showing tumor induction related to PAH exposure.

The development of epidermal papillomas in Lake Ontario fish is likely a co-carcinogenic situation. Recent evidence (Smith and Zajdlik, 1987; Government of Canada, 1991b) indicates that a virus alone may cause tumors. Black's (1983) demonstration that exposure to carcinogenic chemicals will result in skin tumors and the regression of papillomas in fish held in clean water (Smith and Zajdlik, 1987) also imply a chemical factor. The fact that incidence rates appear to be correlated with industrialization and are well above background in those areas implies that a virus and environmental contaminants combined result in more tumors than either is capable of inducing alone.

5.4.5.2 Liver Neoplasms. Two types of liver tumors have been diagnosed in white sucker and brown bullhead populations in Lake Ontario: (1) cholangiolar (bile duct); and (2) hepatocellular. Cholangiolar tumors are generally either discrete nodules of irregularly

arranged ducts within a connective tissue matrix (bile duct adenomas) or extensively invasive, prolific tumors consisting of less well-differentiated, irregular duct-like structures (bile duct carcinomas) (Hayes et al., 1990). In contrast, heptocellular tumors range from small round or irregular foci to larger grossly-visible nodules of phenotypically altered hepatocytes (Hayes et al., 1990). Bile duct tumors are more frequent than hepatocellular tumors in white suckers and brown bullheads from Lake Ontario (Smith and Ferguson, 1985; Hayes et al., 1990; Government of Canada, 1991b).

Surveys of liver neoplasms in white sucker and brown bullhead populations are fairly extensive but tend to concentrate on different regions of Lake Ontario (Figure 5.4.2). The white sucker surveys focus on the north shore whereas the brown bullhead surveys have focused on the Niagara River watershed. The data in Figure 5.4.2 represent total liver tumor rates from single collections made between 1982 and 1987 (Government of Canada, 1991b). These data show that the frequency of occurrence declines moving eastward from the western end of Lake Ontario and the Niagara River, with the exception of a higher than expected occurrence in the Ganaraska River/Cobourg Creek area. This is consistent with the epidermal papilloma data (Figure 5.4.1) and tends to suggest that both types of neoplasm have a common causative factor(s). Reference populations of white suckers at South Bay, Lake Huron, and brown bullheads at Long Point, Lake Erie, both had frequencies of 0% (Government of Canada, 1991b).

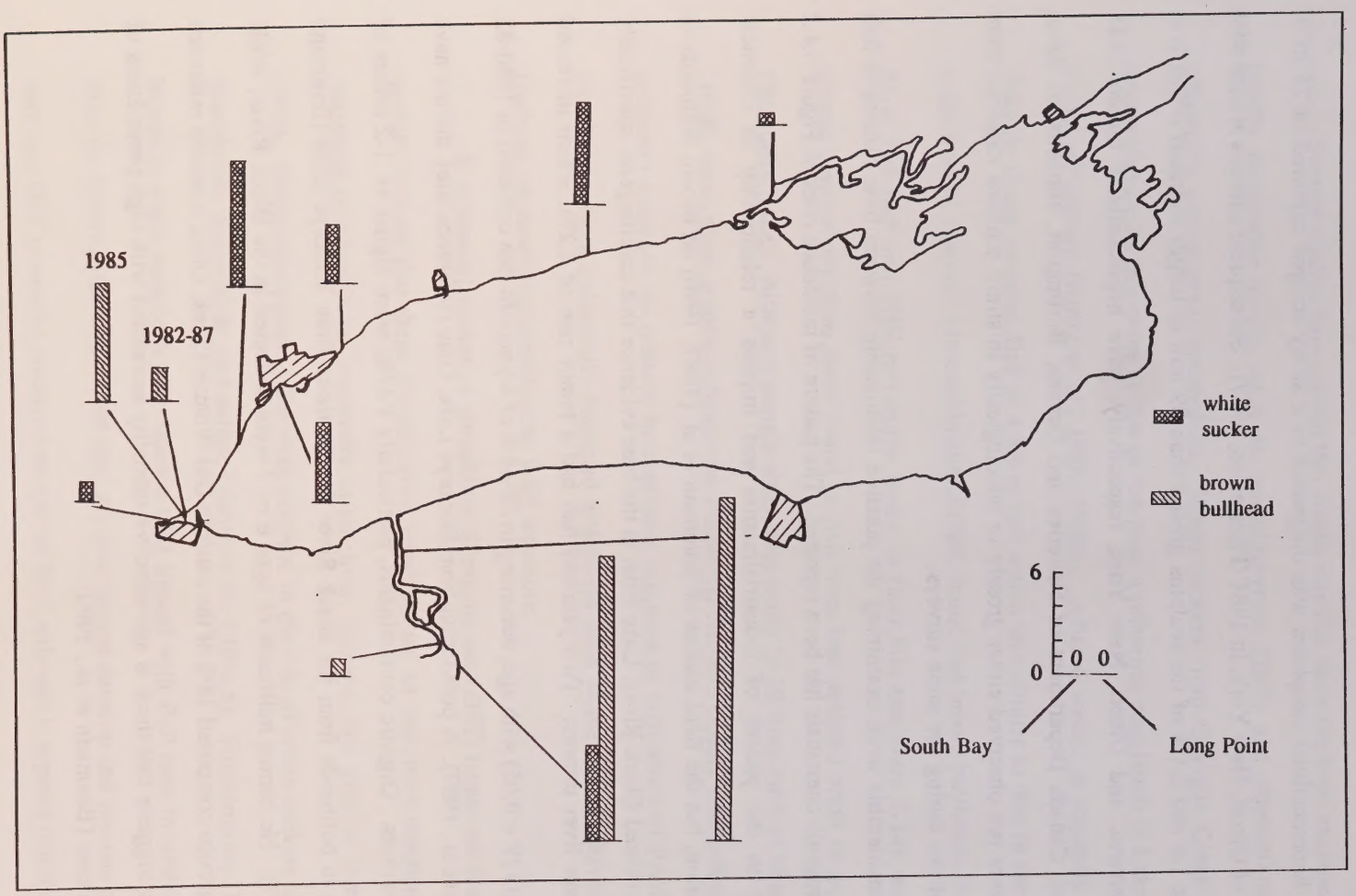

Figure 5.4.2. Prevalence of liver tumors in white sucker and brown bullhead populations in Lake Ontario, 1982-1987.

A hepatocellular neoplasm was diagnosed in a slimy sculpin captured in 75 m of water off of Olcott, New York, in 1984 (Fitzsimons, 1987). Subsequent surveys at nine sites found that 1.8 and 2.1% of the sculpins greater than 89 mm in length captured off of Port Credit, Ontario, and Olcott, New York, respectively, have hepatocellular tumors (J.D. Fitzsimons, Canada Department of Fisheries and Oceans, Burlington, unpublished data). Tumors were not observed either grossly or histologically in slimy sculpins captured from other locations during the same surveys.

Considerable work examining the putative relationship between liver tumors in fish and carcinogenic chemicals has been reported. The pattern of incidence rates in Figure 5.4.2 is similar to the pattern of industrialization and implies a relationship to chemical contamination, but the field studies of Baumann et al. (1987, 1990) on brown bullheads in the industrialized Black River, Lake Erie, is the best evidence that carcinogens, specifically PAHs, cause liver tumors. Two year-old fish had a tumor rate of 1.2%, which increased significantly ($P < 0.05$) with age, occurring in 28-44% of 4-year-old fish collected in 1980-82 (Baumann et al., 1987). A population from Buckeye Lake, Ohio (reference site), did not have any liver tumors. Organic contaminants, particularly PAHs, were higher by 1-2 orders of magnitude in bullheads from the Black River than in those from Buckeye Lake (Baumann et al., 1987). No brown bullheads of ages 6 or 7 were collected in the Black River, while these age groups composed 18% of the catch in Old Woman Creek, Ohio, a nearby reference site, which suggests that there is age-selective mortality associated with high prevalences of liver carcinoma (Baumann et al., 1990).

Several studies have shown that freshwater fish have the liver enzymes necessary to rapidly metabolize PAHs, such as benzo(*a*)pyrene (BaP) and phenanthrene (PHE), to potentially carcinogenic metabolites. White suckers from Oakville Creek excreted BaP metabolites in the bile twice as fast as fish from a reference population in Lake Huron within 7 days of capture (Kirby et al., 1988, 1989b). After 6 weeks in captivity the fish from Oakville Creek excreted BaP at a lower rate which was similar to that in the reference fish at the time of capture. Maccubbin et al. (1988) found that brown bullheads from the Buffalo River had BaP and PHE metabolite levels in their bile that were 7-148 and 7-297 times greater respectively than those found in fish from less polluted areas (e.g., Lewiston and Chatauqua Lake). Although sample sizes were small (5-20 from each of Maccubbin et al.'s (1988) sites), these studies indicate that freshwater fish can rapidly metabolize PAH, that the necessary enzymes are induced by continous exposure to high levels of PAH, and that wild populations from industrially impacted waterways such as the Buffalo River and Hamilton Harbour are receiving considerable PAH exposure.

Biochemical studies on glutathione S-transferases (GST) levels and functions in white suckers indicate that these detoxification enzymes are an important mechanism of cellular resistance to xenobiotics, especially in the liver, kidneys, and gills of fish (Hayes et al., 1990). Orally administered BaP was excreted in the bile of white suckers as conjugates of glutathione, glucuronide, and sulfate (Kirby et al., 1989a,b). Furthermore, GST levels were higher in white suckers from reference sites treated with BaP than in similarly treated fsih from the Oakville Creek polluted site. As liver tumors develop and become more malignant, they lose GST protective mechanisms against BaP, although it appears that this loss must first

be initiated by some other factor such as biliary disease (Kirby et al., 1988, 1989b). These results suggest that PAHs may play a role in the malignant progression stage of hepatocellular tumor development, but only after the cells have lost GST dependent resistance as a result of concurrent factors such as hepatic disease. Note that these results are consistent with the field studies (Baumann et al., 1987, 1990) showing that prevalence increases with age, suggesting a latency period between the beginning of exposure and subsequent tumor development. The studies cited above did not report sample sizes or replication and therefore these results should be considered preliminary until they can be confirmed by other researchers.

Experimental carcinogenesis with contaminated sediments provides the strongest support for the hypothesis that liver tumors in fish are caused by PAHs. Hamilton Harbour sediment extracts induced hepatocellular carcinomas in rainbow trout, at incidence rates ranging from 3.1 to 8.1% depending on the sediment extract "dose", in two separate carcinogenicity assays using sac fry (Metcalfe et al., 1988, 1990). No hepatic tumors were observed in fish treated with sediment extract from Oakville Creek, although some were expected based on the contaminant profile of these sediments, or with an extract from reference sediments in South Bay, Lake Huron. The Hamilton Harbour sediments had the highest levels of PCBs, organochlorine pesticides, and especially PAHS, which were 2-3 orders of magnitude greater than in sediments from the other sites (Metcalfe et al., 1988, 1990). Sediments extracted to enrich for PAH were mutagenic in the *Salmonella*/microsome assay and induced high levels of DNA adducts in cultured mouse cells after in vitro exposure. Two points stand out in these studies: (1) the sediment extract from Hamilton Harbour

induced only heptocellular carcinomas; and (2) these tumors were induced in young fish, in contrast to field observations.

The weight of evidence strongly supports a causative role for PAHs in the development of hepatocellular tumors. Hepatocellular tumors are found only in fish from PAH contaminated sites in the Great Lakes (I.R. Smith, personal communication). Both brown bullheads and white suckers have the necessary enzymes to rapidly metabolize PAHs and excrete carcinogenic metabolites in their bile. However, liver tumors are GST deficient. High PAH levels in sediment extracts likely induced hepatocellular carcinomas in rainbow trout sac fry. Although most of the laboratory studies focus on BaP, a known mammalian carcinogen, the collective weight of evidence from the toxicological profiles of other PAH compounds (see Clement International Corporation, 1990), which are based on other animal studies, is sufficient to conclude that PAHs as a class of chemicals cause heptocellular tumors in fish populations from Lake Ontario. Based on this type of evidence, PAHs must be a strong candidate as the causative agent of hepatocellular tumors in slimy sculpins.

Causation is less clear with respect to cholangiolar tumors. I.R. Smith (personal communication) indicates that bile duct tumors are found at sites contaminated with PAHs, at sites contaminated with other chemicals such as pulp mill wastes, and even at some relatively "clean" sites. Smith and Ferguson (1985) reported some evidence that parasites may be involved in the development of bile duct tumors, although their evidence was inferential based on histological examinations showing cellular changes consistent with the presence of a parasite, but without actually observing a parasite.

5.4.5.3 Gonadal Tumors and Deformities. Gonadal neoplasms were epizootic in carp x goldfish hybrids in the 1970s (Sonstegard, 1977; Dickman and Steele, 1986; Down et al., 1988) and possibly the 1980s (Smith and Ferguson, 1985). Dickman and Steele (1986) reported a correlation between the occurrence of gonadal tumors and industrial loadings in the Welland River. Hybrids captured 3 km downstream from Welland had a rate of 48%, compared to a rate of 12% at a site 10 km downstream, and 0% at a "control" site above Welland. Although the numbers of fish taken in this study were not large (N = 36), the data are consistent with a chemical etiology.

Other evidence suggests that the occurrence of gonadal tumors may not be related to environmental contamination. Sonstegard (1977) also reported that 61% of hybrids captured from the Kincardine area of Lake Huron had gonadal tumors, hence the tumor is widespread, which means that either that the contaminant is widespread (e.g., PCBs or DDT) or that some other causative agent(s) are responsible. Gonadal tumors were rare in the parental species in collections by Dickman and Steele (1986). More recent information is limited and not sufficient to determine the exact etiology or the role of environmental contaminants in development of gonadal tumors.

Ruby and Cairns (1983) examined the histological significance of testicular constrictions among hatchery-reared lake trout stocked in Lake Ontario as yearlings and recaptured when they were 4-7 years old. They examined eight males and found that five had partial or complete constrictions. Histological examinations indicated that the constrictions delayed spermatogenesis cycles throughout the testes at spawning time compared to normal testes. The significance of these observations lies in the irreversibility of the

constrictions and associated changes. Fish in which spermatogenesis is delayed prior to spawning do not initiate a new cycle of spermatogenesis, impairing reproduction. Further work by the DFO indicated that the condition occurs in approximately 30-40% of the male lake trout throughout the Great Lakes, which is 3-20 times greater than in lake trout from inland lakes (Great Lakes Water Quality Board, 1989b). Thus the significance of the condition is not clear at this point. Although Ruby and Cairns (1983) did not establish a specific etiology for the testicular constrictions, environmental contaminants cannot be ruled out. The widespread occurence and relatively higher prevalence in the other Great Lakes tends to suggest some ubiquitous agent(s) are involved.

5.4.6 CONCLUSIONS

The data available concerning fish tumors in Lake Ontario are sufficient to infer that, based on the weight of evidence PAHs are the likely cause of liver tumors in white suckers and brown bullheads in Lake Ontario, particularly in heavily industrialized areas such as Hamilton Harbour. The evidence is not yet sufficient to ascribe a chemical etiology to epidermal papillomas in the same species or gonadal tumors in carp x goldfish hybrids, although chemicals are strongly suspected in both instances. Thyroid hyperplasia (goiters) reported in Pacific salmon do not appear to be related to the presence of toxic substances. The causal factor(s) leading to testicular constrictions in adult male lake trout are not known.

The fish sampled in tumor studies have apparently come from large populations in streams and embayments of Lake Ontario. The distribution and prevalence of tumors in fish populations in Ontario waters are well documented, but there is virtually no information

concerning tumors in fish inhabiting the New York waters of Lake Ontario. Although the presence of tumors might be considered a symptom of "ill-health" in fish, there is comparatively little information with respect to the effects of tumors on the biomass and production of fish populations and communities.

5.5 BIRD OR ANIMAL DEFORMITIES OR REPRODUCTIVE PROBLEMS

5.5.1 DEFINITION

The International Joint Commission's guidelines for use impairment 5, "bird or animal deformities or reproductive problems" are shown below.

Listing Guideline

When wildlife survey data confirm the presence of deformities (e.g cross bill syndrome) or other reproductive problems (e.g eggshell thinning) in sentinel wildlife species.

Delisting Guideline

When the incidence rates of deformities (e.g cross bill syndrome) or reproductive problems (e.g eggshell thinning) in sentinel wildlife species do not exceed background levels in inland control species.

The rationale given for these guidelines is the emphasis on confirmation through survey data and making the necessary control comparisons (IJC, 1991). The delisting guideline is not an exact opposite of the listing guideline. The listing guideline does not require the frequency of occurrence of deformities to be greater than background levels, but merely the presence of deformities. The comparisons with background levels of deformities is only a requirement of the delisting guideline, not the listing guideline. The reason for the difference in listing and delisting guidelines was to make it easy for an area to meet the listing criteria (Tom Coape-Arnold, Ministry of Environment, personal communication).

For the purposes of this section, the definition of deformities was expanded to include biochemical indicators. Biochemical indicators can be sensitive measurements of chemical contamination, and in some instances, can be used to predict biological effects. Biochemical indicators include hepatic mixed function oxidases, hepatic porphyrins, hepatic Vitamin A, thyroid measurements, and genetic alterations. A battery of biochemical tests on an organism can be useful indicators of biological stress. The utility of the biochemical indicator approach is that it is a response to the mixtures of chemicals occurring in the ecosystem, so it overcomes the limitations of one chemical testing. However, the disadvantages of biochemical indicators include the large number of tests available, the degree of sensitivity of a particular test, the difficulty in pinpointing the exact chemical cause and the absence of control populations (Government of Canada, 1991b).

This section will review the available data on bird and animal deformities, reproductive problems and biochemical indicators in Lake Ontario. Over 60 papers were reviewed on 10 species of birds and animals in Lake Ontario. Approximately two-thirds of these papers focused on herring gulls, which reflects the two decades of work done by the Canadian Wildlife Service. The majority of the available information related to the nearshore area rather than the open water. However, many birds forage over a wide range including the offshore area of the Lake, so effects seen in the nearshore colonies reflect a combination of nearshore and offshore causes.

A large number of papers describe contaminant burdens in birds and animals, and fewer articles link these burdens with an effect. Contaminant burdens without a reference to an effect are discussed in Section 5.3, Degradation of fish and wildlife populations. Studies are summarized in Appendix F.

Of the eight RAPS on Lake Ontario, only one, Hamilton Harbour considers that this beneficial use is currently being impaired. The RAPs for Eighteen Mile Creek and Rochester are currently evaluating the applicability of all impairments to the Area of Concern. Many RAPs note the historically high concentrations of organochlorines in wildlife and the occurrence of deformities and reproductive problems. However, most consider the present day situation to be distinct from this historical record.

5.5.2 TYPES OF DEFORMITIES AND REPRODUCTIVE PROBLEMS

Several types of deformities have been documented in wildlife in the Great Lakes:

Birds:

- bill deformities including cross bills
- changes in tarsi length
- enlarged thyroid
- changed thyroid physiology
- reduced embryo size
- enlarged liver
- changed liver physiology
- porphyria
- edema (subcutaneous, pericardial, and peritoneal)
- biochemical alterations
- feminisation
- eggshell thinning
- missing eyes
- defective feathering
- growth retardation
- club feet
- GLEMEDS : Great Lakes Embryo, Mortality, Edema Deformity Syndrome

Animals:

- tail deformities
- missing eyes
- deformed carapace
- deformed jaws and cranium
- hind and limb deformities

Several types of reproductive problems have been documented in the wildlife of the Great Lakes:

Birds:

- low number of young fledged per pair
- high number of eggs lost from the nest
- low nest attentiveness
- reduced thermal control of eggs
- low hatchability of eggs
- absence of nesting
- supernormal clutches
- high chick mortality
- embryonic mortality
- female-female pairing

Animals:

- reproductive failure
- kit mortality
- unhatched eggs

5.5.3 DEFORMITIES AND REPRODUCTIVE PROBLEMS IN HERRING GULLS

The herring gull was chosen as a monitor of spatial and temporal trends in contamination in the Great Lakes in 1972 (Struger at al., 1987). Organochlorine contaminants concentrate in the body fat of the female gull and are transferred along with the fat to the yolk of the eggs. Measuring the contaminants in the eggs provides a snapshot of the contaminant at that time (Braune and Norstrom, 1989). Fish eating birds, such as the herring gull make ideal monitors as they feed at the top of long food chains, stay on the Lakes all year, show little immigration, are easily sampled and are abundant (Weseloh, 1984).

Many of the reports of deformities and reproductive problems were documented in the herring gull populations during the 1970's, and effects have lessened since this time (Gilbertson, 1990, Government of Canada, 1991b). However, the level of scientific effort in

the herring gull research has also decreased since the late 1970's due to funding constraints, and while the contaminant work continues, there is a lack of data on reproductive success or deformities in recent years. The data set on the herring gull contaminants is unique in the Great Lakes as it provides long term contaminant trends taken by experienced researchers, analyzed under strict QA/QC controls with carefully archived samples available for future analysis. This increases the confidence in the trend data, as analytical or sampling artifacts are unlikely to be responsible for the results obtained.

5.5.3.1. Deformities. Deformities have been observed in herring gulls during the regular surveys of Lake Ontario colonies during the 1970's. A chick deformity (inability to fly) was found in only 1 of 850 chicks in Eastern Lake Ontario in 1972 (Gilbertson et al., 1976). Tarsi lengths in chicks were measured in 1973 and 1974 at Scotch Bonnet Island and compared to a control population in New Brunswick. Embryos which pipped but failed to hatch had tarsi 20% shorter than control embryos. The shorter tarsi length was inversely related to liver DDE content. Chicks also showed elevated accumulation of a muco-serous exudate in pectoral muscle abdomen (36%) compared to controls (8%). Only 10% of embryos which hatched had accumulations of this exudate compared to 50% of those that died while pipping (Gilbertson and Fox, 1977). An elevated, but non-significant increase in fluid in the peritoneal cavity was observed in embryos from the Great Lakes compared to controls, and in embryos which died while pipping compared to those that hatched. This elevated level was correlated with PCB concentrations in the liver.

Male gulls were observed to be "feminised" in 1975-1976 (Government of Canada, 1991b). Archived samples of near term embryos and freshly hatched chicks from Lake Ontario colonies in 1975 and 1976 were re-examined and 5 of 7 male chicks had developed ovarian tissue and half the female chicks had abnormally enlarged oviducts (Flint and Verna, 1989 cited in Colborn, 1990). This feminisation is thought to be a result of certain congeners of PCBs, TCDDs and DDT/DDE which can mimic hormones. Internal deformities such as higher thyroid mass, smaller follicle diameter, taller epithelial cells, less colloid and higher prevalence of cell hyperplasia were also observed in 1984 at Scotch Bonnet Island (Moccia et al., 1977).

5.5.3.1. Reproductive success. In the early 1970's, the herring gull population suffered from symptoms similar to chick edema disease, renamed Great Lakes embryo mortality, edema and deformities syndrome (GLEMEDS) (Gilbertson et al., 1991). GLEMEDS is a suite of symptoms including reduced weight gain, subcutaneous and peritoneal edema, swollen liver, porphyria, kidney degeneration and reduced hepatic Vitamin A. Chick edema in poultry was traced to the presence of 1,2,3,7,8,9- hexachlorodibenzo-p-dioxin and subsequently to the ability of the dioxin family to produce edema-like properties. (Cantrell et al., 1969 cited in Gilbertson et al., 1991). Lab studies injecting 2,3,7,8-TCDD showed 20% incidence of embryonic mortality and 40% incidence of chick edema at 100 ppt, and was a potential teratogen (Gilbertson, 1982). When the symptoms of GLEMEDS were seen on Lake Ontario in the early 1970's, analytical techniques did not detect the presence of polychlorinated dibenzo-p-dioxins and dibenzofurans. A later reanalysis of 1974 concentrations in eggs did detect 2,3,7,8-TCDD at approximately 500 ppt, with some

suggestion that they may have been as high as 900 ppt. In 1976 TCDD concentrations had declined to approximately 200 ppt, and further declined in 1977 to 160 ppt. Since 1979, concentrations have remained below approximately 100 ppt (Gilbertson et al., 1991).

Hatching studies confirmed the symptoms of GLEMEDS in herring gull eggs and chicks from Lake Ontario (Gilbertson and Fox, 1977). Recovery began in 1976 and was continued in 1977 (Peakall et al., 1980). Gilbertson applied criteria derived from epidemiology to test the hypothesis that GLEMEDS symptoms were analogous to those associated with chick edema disease and caused by the presence of chick edema active compounds such as planar PCBs, polychlorinated dibenzo-p-dioxins, and/or dibenzofurans. Only a limited number of compounds with a specific planar shape and size can cause GLEMEDS. The evidence supporting the link between GLEMEDS and polychlorinated dibenzo-p-dioxins met all epidemiological criteria of specificity, time order, strength of association, consistency, coherence and predictive performance to varying degrees. Gilbertson concludes that the occurrence of GLEMEDS in Lake Ontario herring gulls during the late 1960s and 1970 was likely caused by the presence of 2,3,7,8-TCDD. However, the high concentrations of PCBs cannot be totally ruled out at this time. This analysis of the herring gull data is among the most comprehensive, and provides a solid basis for determining the effects of organochlorines on gulls.

The reproductive success of herring gulls was very low between 1972 and 1975 on Lake Ontario. All colonies were producing 0.21 young per pair or less, or about one-fifth the "normal" rate of reproductive success of 1.0-1.2 young/pair. The reason for the low success rate was high embryonic mortality and high egg loss (Gilbertson, 1986). The early

mortality of chicks during incubation was 20% in 1973 in Scotch Bonnet Island (Gilbertson, 1974). Some reports put the breeding success in herring gulls in eastern Ontario in 1973 between 0.6 and 0.18 fledged chicks per breeding pair, which was the lowest reported for any North American population. The exact cause of the low reproductive rate could not be linked to any one chemical, rather a number of organochlorine contaminants such as TCDD, PCB and HCB, were elevated at this time.

From research done on herring gulls in Lake Michigan in the early 1960s, both direct embryonic toxicity and behavioral effects on adults were thought to be involved in the low rate of reproduction. This observation was tested by egg exchange experiments carried out between highly contaminated ("dirty") colonies in Lake Ontario and "clean" colonies on the coast and upper Great Lakes in 1975. Embryonic toxicity is determined by the observation of reproductive success with a "clean" adult incubating a "dirty" egg. Behaviorial effects are determined by the opposite combination: a "dirty" adult incubating "clean" eggs (Peakall et al., 1978). Controls are "clean" adult and "clean" egg and "dirty" adult and "dirty" egg. The "clean-clean" pairing gave a 86% hatch, and the "dirty -dirty" pairing gave a 2% hatch. The "clean" birds with the "dirty" eggs (embryonic toxicity) gave a 10% hatch. The opposite combination, "dirty" birds with "clean" eggs (behaviorial effects) gave a 7% hatch. This experiment neatly illustrates that both embryonic toxicity and behavioral effects were influencing reproductive rates in herring gulls. The same experiment was repeated in the two following years (1976, 1977) and showed less clear trends. This was attributed to the dramatically decreasing contaminant concentrations in the colonies during these later years (Peakall et al.,1978). Egg injection experiments could not pinpoint the contaminant

responsible for the embryotoxicity (Gilman et al., 1982). Other workers have confirmed the influence of organochlorines on the behavior of herring gulls: significant differences in nest attentiveness and thermal control in herring gulls from Lake Ontario colonies compared to marine coastal controls (Fox et al., 1978).

A significant correlation still exists between DDE concentrations and eggshell thinning in herring gulls (Weseloh et al., 1990). This relationship was originally presented in the 1970s for a variety of fish eating birds (Teeple, 1977, Gilbertson, 1974). Herring gulls are fairly resistant to DDE induced eggshell thinning compared to other Great Lakes birds (Peakall et al.,1978). DDE concentrations at 5 ppm can cause a 10% reduction in eggshell thickness in herring gulls (Colborn, 1991), which is a greater reduction than seen in many other bird species at this concentration. Observations of flaking and breakage of eggs in herring gull colonies in Lake Ontario in 1972 corresponded with 9 - 16% eggshell thinning, compared to 1 - 6% in Lake Erie (Gilbertson, 1974).

Recent research on herring gulls has focussed on congener specific analysis of PCBs, TCDDs and TCDFs in herring gull eggs. In 1988, the toxic equivalents in herring gull eggs from the Niagara River was 423 ppt compared to 1605 ppt in eastern Lake Ontario. The highest value recorded on the Great Lakes was 3751 at Saginaw Bay. Approximately 85% of the toxic equivalent originated from PCB congener 126, with <5% from TCDD (Norstrom et al., 1989).

There is little data available on reproductive rates in herring gulls in the late 1980s, as the focus of research effort has shifted to biochemical indicators.

5.5.3.3. Biochemical indicators. Five major biochemical indicators have recently been investigated in herring gulls in Lake Ontario: mixed function oxidases, porphyrin, hepatic retinol, thyroid changes and genetic damage. The mixed function oxidases (MFO) contribute to the biological defenses that protect organisms from toxic chemicals. The MFO enzymes add oxygen, making the toxic compound more water soluble and hence more easily excreted (Government of Canada, 1991b). Exposure to PCBs and other contaminants may cause enzyme induction which may lead to the increased or altered metabolism of contaminants. PCBs, PCDDs and PCDFs have been correlated with induction of aryl hydrocarbon hyroxylase (AHH) in experimental rat studies (Safe, 1987 cited in Government of Canada, 1991b). AHH activity has been associated with birth defects, immune suppression, wasting and porphyria in lab animals (Colborn, 1991).

Herring gulls in Lake Ontario in 1981 had significantly elevated levels of one mixed function oxidase and AHH activity in their livers compared to a control population (Ellenton et al., 1985). AHH concentrations in herring gulls from Lake Superior and southern Lake Huron were not significantly different from the control population. Based on these studies, the concentration of MFO enzymes are therefore used as an indicator of exposure to toxic contaminants, and may be indicative of the degree of toxicity of these contaminants. However, MFO can exhibit seasonal, intrasite and individual variation in wildlife and so need to be carefully sampled and compared against control populations (Rattner et al., 1989).

The second biochemical indicator is the size and/or composition of the porphyrin pool. Highly carboxylated porphyrins accumulate in the liver when the normal processes which produce heme are disrupted. Heme is as a building block for haemoglobin and enzymes.

The occurrence of highly carboxylated porphyrins is related to exposure to organochlorines such as PCBs, PBBs, HCB, TCDDs, and steroids (Foley et al., 1988). Embryos from Scotch Bonnet Island in 1974 showed significantly higher porphyrin levels than embryos from control areas (Gilbertson and Fox, 1977). However, these elevated levels were not accompanied by a significant elevation of any organochlorine in the brain or liver. In 1985, concentrations of highly carboxylated porphyrins in livers of adult herring gulls from the Great Lakes were markedly higher than gulls from coastal areas and in seven other species of birds consuming diets uncontaminated with polyhalogenated aromatic hydrocarbons. The highest concentration of highly carboxylated porphyrins were found in gulls from Green Bay, Saginaw Bay and Lake Ontario. While the specific chemical responsible for the elevated porphyrin levels could not be determined, exposure to polyhalogenated polyaromatic hydrocarbons was suspected (Fox et al., 1988). The alterations in porphyrin levels were more severe in 1974 than 1985 (Fox, 1991).

The third biochemical indicator, Vitamin A, is required for vision, reproduction and maintenance of epithelia and mucous secretions. The uptake, storage and mobilization of vitamin A is a highly regulated process which can be altered by a variety of contaminants such as cadmium, PAHs, insecticides, PCBs, PBBs, and TCDDs (Fox, 1991). Nutritional vitamin A deficiency or excess has been shown to elicit changes in secondary sex characteristics, testes weight, spermatogenesis, egg laying, egg size, embryo survival, incubation time, hatchability, and bone and cartilage deformities (Colborn, 1990). Retinol, a form of Vitamin A, can be esterified and stored in the liver or become a carrier protein and distributed to other tissues. Retinol palmitate is the predominant form of stored vitamin A

in vertebrates (Spear et al., 1990).

Concentrations of retinol and retinol palmitate in livers of herring gulls from the Great Lakes in 1982 were significantly lower than a control coastal population. The lowest average concentration of retinol and retinol palmitate came from Presqu'ile colony in eastern Lake Ontario. Retinol and retinol palmitate concentrations were inversely related to the extent of dioxin contamination recorded previously (Spear et al., 1986).

Herring gull eggs from eastern Lake Ontario colonies sampled in 1987 were analysed for retinol and retinol palmitate to evaluate the biochemical effects with present day levels of contamination. The molar ratio of retinol to retinol palmitate in the egg yolk correlated to several indices of PCDD and PCDF concentrations in eggs. A PCB congener, 3,3',4,4', caused a significant increase in the molar ratio in ring dove eggs in a lab experiment. This indicates that a specific PCB congener may alter yolk retinoid concentration. The sensitivity of retinol as a biomarker is also seen: retinol concentrations were affected at concentrations that did not cause the accumulation of porphyrin in the liver (Spear et al., 1990).

Sixty nine percent of the samples from Lake Ontario herring gulls had concentrations of retinol less than 100 μg/g compared to the control population in which 85% of the samples had retinol concentrations over 100 μg/g (Fox and Trudeau, cited in Government of Canada, 1991b). The concentrations of stored retinols in Lake Ontario were lower in 1985 than in 1982. A 50% decline in retinol palmitate concentrations in the livers of herring gulls on Lake Ontario was observed between 1985 and 1987 (Fox and Trudeau, cited in Colborn, 1989b). This could indicate continued and possible increased interference in Vitamin A processes. This supports the hypothesis that present day concentrations of chlorinated dioxins and

possibly other contaminants such as PCBs are affecting Vitamin A stores in herring gulls in the Great Lakes. This evidence also provides support for considering PCBs a causal pollutant in Lake Ontario.

The fourth biochemical marker is the size and/or condition of the thyroid gland. The thyroid gland is an initiator, regulator and moderator of a number of processes in vertebrates. Enlargement of the thyroid and structural cellular changes may indicate exposure to toxic chemicals (Government of Canada, 1991b). The thyroid mass of herring gulls from eastern Lake Ontario in 1985 was almost double the control population, and also exhibited a 50% increase in epithelial hyperplasia (Fox cited in Government of Canada, 1991b). The degree of thyroid enlargement and cell hyperplasia is significantly correlated with liver PCB concentrations (Fox and Moccia, cited in Government of Canada, 1991b). Similar to the trend observed for retinol levels, the severity of goiter in herring gulls is also increasing. Free plasma iodine was lower, but not significantly lower than in the control populations. This suggests that herring gulls in Lake Ontario and in the Great Lakes suffer from persistent thyroid stimulating hormone activity resulting in goiter, rather than iodine deficiency. This evidence provides support for two criteria for a cause-effect relationship: specificity of the association, as a very specific effect, goiter, is observed with elevated PCB concentrations and strength of association, as a more intense exposure resulted in a more severe effect.

The fifth biochemical marker, changes in genetic material causing alterations in DNA, can be caused by exposure to toxic chemicals from the Great Lakes. No significant difference was observed between the rate of sister chromatid exchange in herring gull embryos from the Great Lakes when compared to a control population in 1981 (Ellenton and

McPherson, 1983). The extracts of the herring gull eggs from all Great Lakes samples were also not mutagenic in the *Salmonella*/microsome test, but the extract for Lake Ontario eggs produced a significantly higher frequency of chromosomal aberrations at one quarter the dose of the other samples. However, a strict dose-response relationship was not observed (Ellenton and McPherson, 1983).

In two other tests of changes in genetic material in herring gulls from Lake Ontario, the number of DNA strand breaks in herring gulls in 1987 was not significantly different from a control population (Shugart and Peakall, unpublished data cited in Government of Canada, 1991b). The second test found a significantly lower percentage of 5-methyl deoxycytidine in the livers of herring gulls in 1987 compared to a control population. This indicates a lack of methylation of DNA, which may be caused by chemical contaminants (Shugart and Peakall, unpublished data cited in Government of Canada, 1991b). These genetic studies were initiated after the peak of chemical contamination in the mid 1970's, and so cannot provide information on the genotoxicity of the chemical concentrations at that time.

These biochemical markers provide useful evidence on the effect of contaminants on herring gulls in Lake Ontario from the 1970's to the present day. Taken with other studies on herring gulls, the evidence provides support for three of the criteria used to establish a cause-effect relationship: consistency of observations, as herring gulls, in different Great Lakes showed similar effects; strength of association as more intense exposure in Lake Ontario produced more severe effects; and coherence with other information, as numerous studies have found similar results. This body of evidence can therefore support the cause-effect relationship between polyhalogenated, polyaromatic hydrocarbons such as PCBs and

dioxins and deformities in wildlife species such as herring gulls.

5.5.4 DEFORMITIES AND REPRODUCTIVE PROBLEMS IN RING-BILLED GULLS

Much less information on deformities and reproductive problems is available for the ring-billed gull compared to the herring gull. In 1972 and 1973, a leg deformity (perosis) was found in about two dozen chicks in the Toronto colony. This deformity prevented the birds from being able to stand or walk properly. One abnormal chick with a bill deformity was found at Pigeon Island in 1973, and no further deformities were found in over 500 chicks from 4 colonies in the Great Lakes in 1972-1973. Mean PCB concentrations in eggs in Lake Ontario were elevated (379 ppm) (Gilbertson et al., 1991). The ring-billed gull experienced two major die-offs in Southern Ontario in 1969 and 1973. Examinations of contaminant concentrations in the brains of dead birds revealed concentrations of PCBs, DDE, and dieldrin 30-90 times higher than living gulls (Sileo, 1977). It is possible that these elevated concentrations of contaminants resulted in the death of ring-billed gulls.

5.5.5 DEFORMITIES AND REPRODUCTIVE PROBLEMS IN COMMON TERNS

5.5.5.1. Deformities. Common tern colonies in 1972 and 1973 contained the greatest proportion and widest variety of chick abnormalities in birds examined. Deformities included crossed bills, abnormal mandibles and supernumerary toes at the tarsometatarsal joint. Cross-bill deformities or other gross defects occurred in 11 of 844 common tern chicks in Lake Ontario in 1971-1973, compared to 2 of 1733 chicks at Port Colbourne on Lake Erie

(Gilbertson et al., 1976). This is the highest rate of deformities ever recorded in any species on the Great Lakes. Common terns appear to be very sensitive to chemical contamination. Concentrations of contaminants such as PCBs and DDE in common terns in Hamilton Harbour in 1971 increased over the season. The arriving birds had lower concentrations of contaminants, and concentrations increased with the time spent in the Harbour. This suggests a local source of these contaminants. Mercury concentrations did not show this increase in concentration over the season (Gilbertson, 1974).

In Eastern Lake Ontario colonies in 1972-1973, chick abnormalities were less than 1% and were associated with increased PCB concentrations. In areas with no abnormalities, the PCB concentration was approximately 155 ppm, and in areas of high abnormalities the PCB concentration was 257 ppm (Gilbertson et al., 1976). The frequency of deformities in common tern colonies in Lake Ontario has returned to background levels (Fox, personal communication, cited in Peakall et al., 1980).

5.5.5.2. Reproductive Problems. The common tern experienced poor rates of reproduction in 1972, when rates were significantly lower than Lake Erie (Morris et al., 1976). The main causes of reproductive failure were poor egg hatchability and the disappearance of eggs and chicks. Eggshell cracking and breakage was also observed. The reproductive success on Gull Island in 1976 was also lower than in Lake Erie and other Lake Ontario habitats (Morris et al., 1980). However, this study employed highly intrusive methods of management of the colony which alone could have contributed to the reproductive problems.

5.5.6 DEFORMITIES AND REPRODUCTIVE PROBLEMS IN CASPIAN TERNS

5.5.6.1. Deformities. The only record of deformities in Caspian terns found abnormalities in the chicks at a rate of 1 per 100 in 1972-1973 at Pigeon Island. PCB concentrations were 359 ppm (Gilbertson et al., 1976).

5.5.6.2. Reproductive Problems. There are no comprehensive records of reproductive rates in Caspian terns in Lake Ontario. Most of the data on this species is gathered on Lake Huron and Lake Michigan.

5.5.7 DEFORMITIES AND REPRODUCTIVE PROBLEMS IN BLACK-CROWNED NIGHT-HERON

5.5.7.1. Deformities. At Pigeon Island in 1972, no deformities were found and in 1973, 1 out of 39 chicks of black-crowned night-heron examined had a deformity (crossbill). The PCB concentration was 304 ppm (Gilbertson et al., 1976).

5.5.7.2. Reproductive Problems. In 1969, there was close to total reproductive failure in black crowned night-herons at Pigeon Island. In 1972, the reproductive rate was 0.22 young/pair. The geometric means for DDE ranged from 4 -12 ppm, dieldrin 0.2-0.5 ppm, and PCB from 10-62 ppm in 1972 -1976. In 1982, these levels had decreased to 1.6-4.8 ppm DDE, 0.07-0.31 ppm dieldrin, and 19-32 ppm PCB. Eggshell thinning rates of 14-16% were observed in this colony from 1972 -1976. In 1982 the rate of thinning had decreased to 2.5 to 8% (D.V. Weseloh, personal communication, cited in Peakall, 1990). The eggshell thinning was linked to elevated concentrations of DDE (Government of Canada, 1991b).

5.5.8 DEFORMITIES AND REPRODUCTIVE PROBLEMS IN DOUBLE-CRESTED CORMORANTS

5.5.8.1. Deformities. The incidence of deformities in chicks of double-crested cormorants was 3.5 per 10,000 from 1979 -1987 in Lake Ontario. This is higher than the deformity rate of 2.4 and 0.6 deformities per 10,000 in control populations and 0 deformities on Lake Erie (Fox et al., 1991b). The deformities were associated with elevated levels of PCBs.

5.5.8.2. Reproductive Problems. Large declines in cormorants populations were noted on the Great Lakes in the 1960s, and 1970s and were attributed to breakage and disappearance of eggs. In 1972, 95% of the eggs were lost or broken at the end of the incubation period on the Great Lakes (Weseloh et al., cited in Peakall, 1990). In Eastern Lake Ontario from 1978 to 1982, DDE concentrations in eggs decreased 50% and eggshell thinning decreased from 24% to 6% (Price et al., 1986). No cormorants are known to have fledged from any colonies in the Canadian waters of Lake Ontario from 1954 to 1977 (Fox et al., 1991a). Cormorant populations have shown a dramatic increase with reproductive rates of 1.4 -1.5 young/ nest in Lake Ontario in 1981 (Weseloh et al., 1983).

In a study to determine genetic damage, the number of DNA strand breaks in liver samples in young cormorants in 1987 in Lake Ontario was not significantly different than in young cormorants from the Atlantic coast (Shugart and Peakall, unpublished data, cited in Government of Canada, 1991b). Older cormorant chicks collected in 1988 in Lake Ontario also did not show elevated levels of mean EROD activity compared to a control population (Peakall, unpublished data, cited in Government of Canada, 1991b).

5.5.9 DEFORMITIES AND REPRODUCTIVE PROBLEMS IN BALD EAGLES

5.5.9.1. Deformities. There are no records of deformities in bald eagles in Lake Ontario.

5.5.9.2. Reproductive Problems. Historical records indicate that bald eagles once nested along the shores of Lake Ontario. However, in 1969 only 1 of the 12 nests in Eastern Ontario produced young (Government of Canada, 1991b), and in 1974 no nests produced young (Weekes, 1975). This decline in reproductive success was observed elsewhere on the Great Lakes, particularly along Lake Erie. There were no nests along Lake Ontario in the 1980s.

The Ecosystems Objectives Working Group is developing indicators of ecosystem health for the Great Lakes. One of the suggested indicators is the bald eagle. For example, along Lake Ontario, a healthy ecosystem would have approximately 20 pairs of bald eagles with a reproductive rate of 1 chick per pair. The maximum contaminant concentrations in the eggs to achieve these rates are also specified (Gilbertson, 1989b).

Colborn (1991) has applied the same epidemiological criteria used by Gilbertson et al., (1991) to test the hypothesis that organochlorine chemicals are the cause of reproductive loss in the Great Lakes Basin. Shoreline birds bear significantly higher concentrations of persistent organochlorines than inland birds. Maternal prezygotic exposure was thought to affect the viability of embryos and chicks. This was supported by the use of a fledging ratio: the mean number of fledglings per successful territory to the mean number of fledglings per active territory. When the numerator is greater than 1.4, exposure of parental birds and offspring to toxic chemicals may be a factor. When the ratio is greater than 2, parental

physiological changes is underway.

Mink containing biologically significant body burdens of these contaminants have been found along the Lake Ontario shoreline. Livers of 5 minks on the northern shore of Lake Ontario contained 21 ppm on a lipid weight basis (D.V.Weseloh, unpublished data, cited in Government of Canada, 1991a). However, high concentrations of PCBs in mink livers were also found away from the shore. Concentrations in Prince Edward County were 3 ppm, 8 km from the shore, while the St.Lawrence area had significantly higher PCB concentrations in mink livers closest to the shore. In 1982-1984 in Oswego County, on the south side of the Lake, mink livers contained 0.3 ppm PCB and 7 ppm in fat tissue (Foley et al., 1988). PCB concentrations in mink were compared to fish PCB concentrations. On a restricted geographic scale, PCB concentrations in mink were significantly correlated with PCB concentrations in fish. The difficulty in establishing effects of PCBs on natural populations of mink include the number of other factors which influence mink reproductive success such as flooding, habitat, presence of muskrats, and trapping intensity. Therefore, trying to tease out the effects of contaminants on the reproduction of mink is a difficult process.

Wren (1991) has applied the five criteria derived from epidemiology to test the hypothesis that wild mink which are exposed to PCB and dioxin through the consumption of Great Lakes fish exhibit reproductive dysfunction and population declines. The data relating to time order were indeterminate, while the strength of association between chemical exposure and population effects supported the hypothesis. The third criterion, specificity of effects of PCBs and dioxins on mink reproductive and mortality rates is strongly established in experimental studies, but less well established in field situations. The fourth criterion,

consistency, supported the hypothesis. Theoretical and biological coherence factors also supported the hypothesis with factual and statistical coherence factors being indeterminate.

Therefore, studies on ranch mink fed PCB contaminated food confirm the sensitivity of mink to PCBs and lend support to the suspicion that PCBs are responsible for the decline in mink populations in the Great Lakes.

5.5.12 DEFORMITIES AND REPRODUCTIVE PROBLEMS IN SNAPPING TURTLES

5.5.12.1. Deformities. Information on deformities in common snapping turtles comes from surveys in 1986-1989 of seven sites, three of which are in the western basin of Lake Ontario; Cootes Paradise in Hamilton Harbour, Cranberry Marsh, and Lynde Creek. The study compared the reproductive rate and deformity frequency from artificially incubated eggs from these sites to populations in Algonquin Park. The percentage of deformed hatchlings in Lake Ontario sites was significantly higher than the relatively "clean" site at Algonquin Park. The percentage of deformed hatchlings in the Lake Ontario sites was from 5 to 40% compared to the 2.6 to 5.0% at the Algonquin site (C. Bishop, unpublished data, cited in the Government of Canada report, 1991b).

Deformities found in hatching turtles and embryos included deformed tails, deformed hind and fore limbs, enlarged yolk sacs, missing eyes, deformed carapaces, deformed craniums (including deformed nostrils), and deformed upper and lower jaws. Bent, curled, and stubby tails were the most common abnormality in all populations. The second most common abnormality was deformed hind limbs, which were stunted and sometimes rotated inward. Sites with the highest contamination also had the highest rate of deformities. There

was a significant correlation between abnormalities and individual PCB congeners and the sum of all congeners. The only compound that was significant in all years and sample types was 2,3,3',4,4'-pentachlorobiphenyl. Pesticide concentrations were not significantly correlated to abnormalities in all years and sample types, although t-nonachlor and chlordane were significant in 1987-1988.

5.5.12.2. Reproductive Problems. This study also found that the percentage of unhatched eggs was significantly higher in the Lake Ontario sites compared to the Algonquin site. The percentage of unhatched eggs at the Lake Ontario sites ranged from 10 to 45% compared to 4 to 19% in Algonquin populations (C. Bishop, unpublished data, cited in Government of Canada, 1991b).

The same five criteria derived from epidemiology that were used by Mac and Edsall (1991) Gilbertson et al. (1991) Colborne (1991) and Wren (1991) were also applied to the data from the snapping turtle surveys. The hypothesis tested was that elevated incidences of egg death and/or deformities of hatching turtles would occur in populations with high concentrations of organochlorine contaminants. The significantly higher percentage of deformities and unhatched eggs in Lake Ontario wetlands combined with the data indicating significantly higher levels of PCBs, dibenzo-p-dioxins, and dibenzofurans in two of the three sites on Lake Ontario provided support for the criteria of strength of association. However, the evidence for time order and specificity was not sufficient to support the hypothesis. Consistency among studies and the biological and statistical coherence factors supported the hypothesis. The factual and theoretical coherence factors were indeterminate to the hypothesis. Therefore of the five criteria used to evaluate cause-effect linkage, two criteria,

strength of association and consistency, support the hypothesis and two criteria, time order and specificity, have an indeterminate effect on the hypothesis, but do not detract from it. The fifth criteria, coherence, has two factors, biology and statistics, which support the hypothesis and two factors, theory and factual, which have an indeterminate effect on the hypothesis (Bishop et al., 1991a).

5.5.13 CONCLUSIONS

A variety of deformities and reproductive problems have been described in a number of species in Lake Ontario in the 1970s and 1980s. These symptoms are supported by laboratory studies on toxicology and on the results of feeding Lake Ontario fish to ranch mink. Noticeable species differences in deformities and reproductive problems are evident in Lake Ontario. For example, herring gulls had the highest concentrations of contaminants but the lowest rate of chick abnormalities compared to ring-billed gulls, black-crowned night-heron, common, and Caspian terns (Gilbertson et al., 1976). Despite the differences in species sensitivities to organochlorine contaminants, many species showed similar deformities and low reproductive success. For example, the deformities in the fore and hind limbs of snapping turtles are similar to the deformities in the supernumerary digits at the metatartsial joint in fish-eating birds. The GLEMEDS suite of symptoms has been observed in a variety of birds in the Great Lakes.

Distinct time trends are also apparent. The severe reproductive problems in herring gulls and other fish-eating birds in Lake Ontario in the 1970's associated with high concentrations of organochlorines can be contrasted to the lower levels of organochlorines

in the late 1980s.

Many of the symptoms of exposure to PCBs, dioxins, and furans are similar, making causal determination difficult. In addition to having similar symptoms, PCBs, dioxins, and furans are often found together in birds and animals. This co-occurrence makes separating the effects of one contaminant from another very difficult (Colborn, 1991).

From this impairment, the following contaminants should be listed as causal pollutants on Lake Ontario: DDT/ DDE as seen in bald eagle and herring gull populations; PCB from bald eagle, herring gull, and snapping turtle populations; and TCDD and TCDF from herring gull and snapping turtle populations.

5.6 DEGRADATION OF BENTHOS

5.6.1 DEFINITION

The International Joint Commission's guideline for use impairment 6, "degradation of benthos" are as follows below.

Listing Guideline

When the benthic macroinvertebrate community structure significantly diverges from unimpacted control sites of comparable physical and chemical characteristics. In addition, this use will be considered impaired when toxicity (as defined by relevant, field-validated, bioassays with appropriate quality assurance/quality controls) of sediment-associated contaminants at a site is significantly higher than controls.

The rationale for this guideline is given as accounting for community structure and composition, recognising sediment toxicity, and using appropriate control sites (IJC, 1991). The guideline has both the ecological component of measuring community structure and the toxicological component of bioassays. The guideline also has a strong statistical basis inferred from the need to establish that the community structure significantly diverges from an unimpacted control site. An area may be delisted with respect to degradation of benthos by the following guideline.

Delisting Guideline

When the benthic macroinvertebrate community structure does not significantly diverge from unimpacted control sites of comparable physical and chemical characteristics. Further, in the absence of community structure data, this use will be considered restored when toxicity of sediment-associated contaminants is not significantly higher than controls.

Contaminant concentrations in the bottom sediments of Lake Ontario are the result of a number of processes including sorption, advective and diffusive mixing, sedimentation – resuspension, burial and bioturbation. Lake Ontario has four sedimentation basins : Niagara, Mississauga, Rochester and Kingston Basins. Soft silty clays and fine grained sediment accumulates in these zones. The Kingston Basin is often considered a separate sedimentation unit from the other three (Frank et al., 1979). These basins are separated by nondepositional sills (Figure 5.6.1). Sediments in the inshore zone around the periphery of the lake and on the three cross lake sills consist of lag sands and gravels overlying glacial till and glaciolacustrine clays. Occasional outcrops of bedrock are found in this zone (Frank et al., 1979).

Sediment resuspension is a major process in the redistribution of sediments among these basins in the nonstratified period, and the benthic nepheloid layer can be a significant factor during the stratified period (Stevens, 1988). The nepheloid layer is probably a layer of high turbidity and low transmittance caused by resuspended bottom sediments. The thickness of this layer varied from 22 m to 45 m in 1981 (Oliver and Charlton, 1984). Current speeds greater than 1 cm/sec exist in the bottom waters of Lake Ontario throughout most of the stratified season. Thus, the nepheloid layer is probably a significant factor in the distribution of sediments and hence contaminants.

Many authors have measured sedimentation rates for Lake Ontario, which range from 80 to 1220 g/m^2/ year (Nriagu et al., 1983). The sedimentation rate in Lake Ontario is approximately one-seventh that of Lake Erie (Frank et al., 1979). Major differences exist between the sedimentation rates in the nearshore and offshore zones. The sedimentation rates

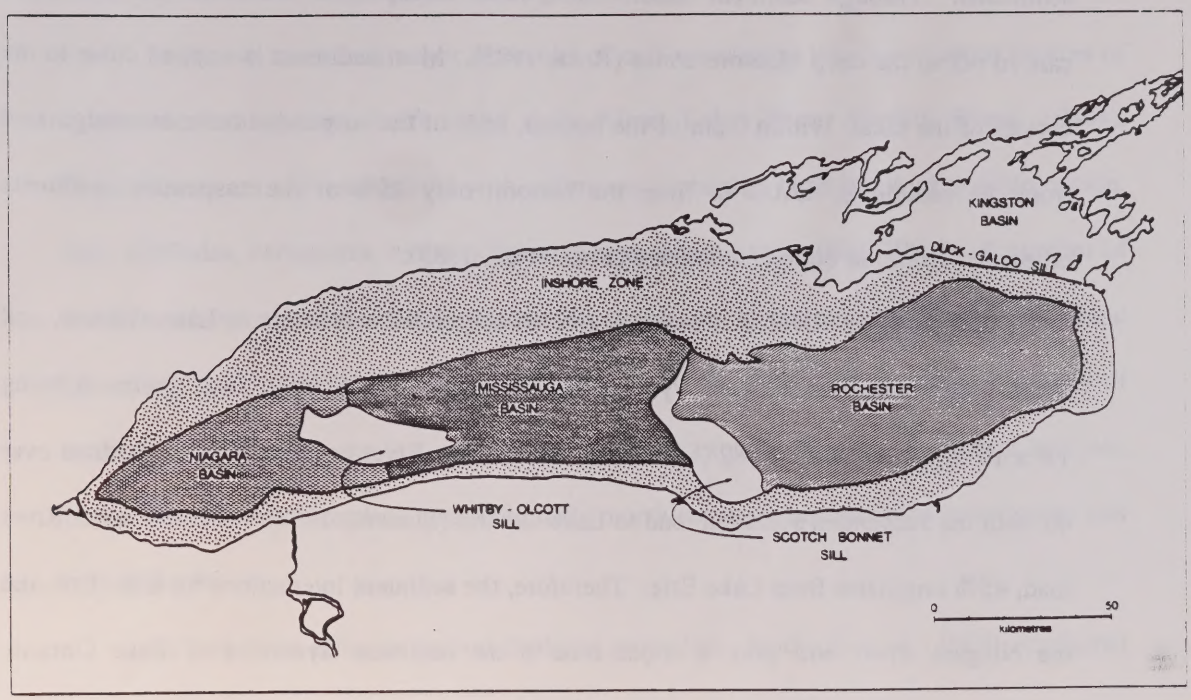

Figure 5.6.1. Basins of sedimentation in Lake Ontario.

Source: Thomas (1983).

at 5 m above the lake bottom in the nearshore zone are considerably higher than the offshore zone. Sedimentation rates in the nearshore zone are thought to be controlled by particulate concentration in the epilimnion and in the offshore zone by sediment resuspension (Rosa, 1985). Sedimentation rates can vary greatly from week to week. From March to November, 1981, minimum to maximum sedimentation rates in the nearshore zone varied by 40, and in the offshore zone by a factor of 10 (Rosa, 1985). Shallow nearshore areas are prone to storm events and so tend to act as a continuous and highly variable mixing and dispersal zone for

sediments. Through sediment translocation, these resuspended sediments are eventually carried off to the deep offshore zones (Rosa, 1985). Most sediment is trapped close to the bottom of the lake. Within 0.2m of the bottom, 85% of the suspended sediments originated from the sediments. At 7 m from the bottom, only 25% of the suspended sediments originated from the bottom sediments (Rosa et al., 1983).

The Niagara River is a major source of suspended sediments to Lake Ontario, and loads approximately 4.56 x 10^6 t/yr (Kemp and Harper, 1976), with other estimates being 1.8 x 10^6 t/yr from 1979 -1982 (Lum et al., 1987). The River is estimated to contribute over 40 % of the suspended sediment load to Lake Ontario (Stevens, 1988). Of the Niagara River load, 45% originates from Lake Erie. Therefore, the sediment interactions on Lake Erie and the Niagara River will play a major role in the sediment dynamics of Lake Ontario. Approximately 94% of the suspended sediments entering Lake Ontario were predicted to be retained in the depositional basins (Lum et al., 1987).

Biological factors as well as sedimentation can play a major role in contaminant transport. In a study of mirex transportation from Lake Ontario into the St.Lawrence, migrating eels were estimated to transport twice as much mirex out of Lake Ontario as suspended sediments (Lum et al., 1987). Bioturbation of the sediments by benthic organisms can resuspend contaminants.

Three general methods are used to assess the degradation of benthos: analysis of sediments for contaminants; comparisons of benthic community structure between contaminated and uncontaminated areas; and sediment bioassays. In this section, each of these methods and the data arising from these methods for Lake Ontario will be discussed.

5.6.2 SEDIMENT CONTAMINATION

Sediments can act as major reservoirs of contaminants in the Great Lakes, sources of continued contamination, buffers and sinks (Great Lakes Water Quality Board, 1989a). Contaminants are often associated with sediments containing high levels of organic carbon, clay, sulphides, carbonates, hydrous iron and manganese oxides (Sly, 1984). A number of data bases have recorded concentrations of contaminants in bottom sediments of the Great Lakes, and more particularly in Lake Ontario. These data bases have employed a variety of methods of sediment sampling, sample handling and extraction, and contaminant analysis. These differences in sampling and analytical procedures make comparisons among data sets very difficult.

As part of an effort to review dredging guidelines, Mudroch et al. (1988) conducted a literature review to obtain recent and background concentrations of selected elements in sediments from the Great Lakes. The large differences in sampling localities, techniques and analytical methods resulted in large differences in concentrations found in the sediments. There was some agreement in the concentration of pre-colonial sediments, but wide differences in concentrations recorded in surface sediments in depositional basins. The authors urged the development of a common protocol for sampling and analysis to provide a comprehensive picture of sediment contamination in the Great Lakes. This concern for standardized protocols and quality assurance/quality control has been echoed by the Great Lakes Water Quality Board (1989a). Keeping these limitations in mind, the concentration ranges for selected heavy metals in the Great Lakes are listed in Table 5.6.1.

Table 5.6.1. Concentrations of elements in sediments of Lake Ontario.

	Reported ranges of concentrations in sediments (μg/g) of Lake Ontario				
	Arsenic	Cadmium	Mercury	Nickel	Zinc
Depositional Basins:					
– surface	0.2-17.0	0.1-6.2(4)	0.140-3.95(3)	29.0-99.0(3)	87-3,507(3)
– background	-	0.9-3.7(2)	0.30-0.9(1)	42-48(1)	83-163(2)
Nondepositional Zones:					
– surface	0.2- 24.0	0.20-20.5(5)	0.01-7.76(8)	4.0-160(5)	6-1,120(9)
– background	-	-	0.40-0.70(1)	.30-100(1)	100(1)
Embayments:					
– surface	-	0.30-22.0(1)	0.01-1.20(1)	-	14-1,225(1)
– background	-	-	-	-	-
Harbours:					
– surface	-	0.5-10.0(4)	0.01-7.00(6)	1.0-75.0(5)	5-2,010(6)
– background	-	1.1[1](1)	-	14[1](1)	210[1](1)
River Mouth:					
– surface	-	1.2-18.3(3)	0.01-3.90(3)	8.9-140	24.5-500(2)
Bluffs:	-	0.2-3.2[2](1)	0.01-.80(1)	-	16-65[2](1)
Soils:	-	-	-	-	-

(n) = Number of surveyed reports
[1]Mean, number of samples = 90
[2]Northern and southern shores of Lake Ontario

Source: Mudroch et al., (1983).

Sediment quality is often assessed by comparison with sediment quality guidelines used for dredging. There are two main sets of guidelines, the Ontario Ministry of Environment's (OME) Guidelines for the disposal of sediment in open water, and the U.S. EPAs guidelines for dredging. These guidelines are discussed more fully in Section 5.7, Restrictions on Dredging Activities. Concentrations of metals regularly exceed guidelines in harbours in Lake Ontario. In an 1981 survey of nearshore zones in Lake Ontario, PCBs were found in 14 of 39 sediment samples. The western shore of the Kingston Basin had the highest contaminant concentrations in the survey. In Port Hope, copper, lead, iron, zinc, chromium and nickel exceeded OME guideline concentrations in the inner harbour (Hart et al., 1986). Uranium concentrations were also elevated in the turning basin. Toronto Harbour sediments exceed OME guidelines for metals and PCBs (Munawar et al., 1989). Hamilton Harbour has among the highest zinc concentrations in lake sediments across Canada. Zinc concentrations range from 5000 to 6000 μg/g in 1982, compared to Toronto which ranged from 300-400 μg/g (Nriagu et al., 1983). Elevated concentrations of heavy metals also occur throughout the harbour, and go down to 60 cm depth vs 20 cm depth in the midlake. PCBs and organic insecticides are also high in the Harbour. Oswego Harbour has elevated mirex concentrations in sediments with values as high as 1800 ng/g close to the source, Armstrong Cork Company.

The Niagara River has elevated concentrations of a number of metals and organics in the sediment; however, as the flow rate through the Niagara River is so great, there are few areas of sediment deposition. In an 1981-1983 survey, all attempts to remove an undisturbed sediment core were unsuccessful (Mudroch and Duncan, 1987). Surface sediment samples

sediments exceeded OME guidelines at all 8 stations in the Niagara River, and the concentration of lead exceeded guidelines at 5 of the 8 stations (Mudroch and Duncan, 1987). The concentration of heavy metals was greatest in the small-sized particles (<13 μm). Elevated concentrations of chlorobenzenes are found in the sediments, and of particular interest is the presence of two fluorinated compounds which have been linked to leaking dumpsites along the Niagara River. These two compounds have been found up to 100 km away in the sediments of Lake Ontario (Kaminsky et al., 1983). The Bay of Quinte had a mean PCB concentration of 48 $\pm$ 43 ng/g in surficial samples in 1972-1973 (Frank et al., 1980). No difference in concentration was found between areas of sedimentation and areas of nonsedimentation. The Trent River and Moira River were identified as active sources of PCBs to the Bay.

Sediment samples were collected from 25 transects on the Canadian nearshore zone in 1981 for the Ontario Ministry of Environment. These samples were analyzed for chemical contamination and benthic invertebrate abundance. Heavy metal concentrations along the south shore transects were not elevated but local oil and grease contamination was noted. The Burlington bar showed an increase in heavy metals and PCBs. Between Oakville and Clarkson, the sediment was heavily contaminated by PCBs, metals, and oil. Silt at 20 m depth off the Toronto islands had elevated PCBs and heavy metals. Sediment along the north shore was characterized as uncontaminated with the exception of PCBs, HCBs and DDE near Ajax. In the sheltered waters of Prince Edward Bay, high concentrations of PCBs, HCB, DDE, mirex, BHC, chlordane, dieldrin, and eldrin were found (Wilkins, 1985).

In the surveys of sediment done in the offshore zones of Lake Ontario, chlorobenzene concentrations are among the highest in the Great Lakes (560 ppb in 1980) (Oliver et al., 1982). The composition of chlorobenzenes also differed: in Lake Ontario, both common and unusual chlorobenzenes were found, while other Great Lakes appeared to lack the unusual chlorobenzenes (Oliver et al., 1983). Chlorobenzene loadings in the sediment of Lake Ontario are closely matched to the history and production of this chemical. Most chlorobenzenes entered Lake Ontario after 1941. The sources of chlorobenzenes are the dumpsites along the Niagara River such as Hooker, and the Oswego River (Oliver et al., 1982). Sediment cores show some reduction in chlorobenzene loadings in the uppermost recent layer (Oliver and Nicol, 1982).

Sediment cores clearly show association between chemical production of PCBs and their concentration in the sediment of Lake Ontario and the historical trend of contaminant loadings in Lake Ontario (Figure 5.6.2). Cores also show that the concentration of Hg, Pb, Zn, Cr, and Cu are decreasing in the open lake sediment (Mudroch, 1983). The influence of the Niagara River is clearly seen in the trend towards increasing concentrations of most pollutants at the mouth of the River. Mercury concentrations in the surface sediments of most of the Lake were less than 500 ng/g in 1968 and 1974, whereas mercury concentrations at the mouth of the River were 5000 ng/g. Similarly with PCBs, concentrations in most of the open lake sediments were less than 100 ng/g and at the Niagara River were 300 ng/g. Mirex concentrations in most of the open lake were below detection limits, but over 10 ng/g at the mouth of the River (Thomas, 1983).

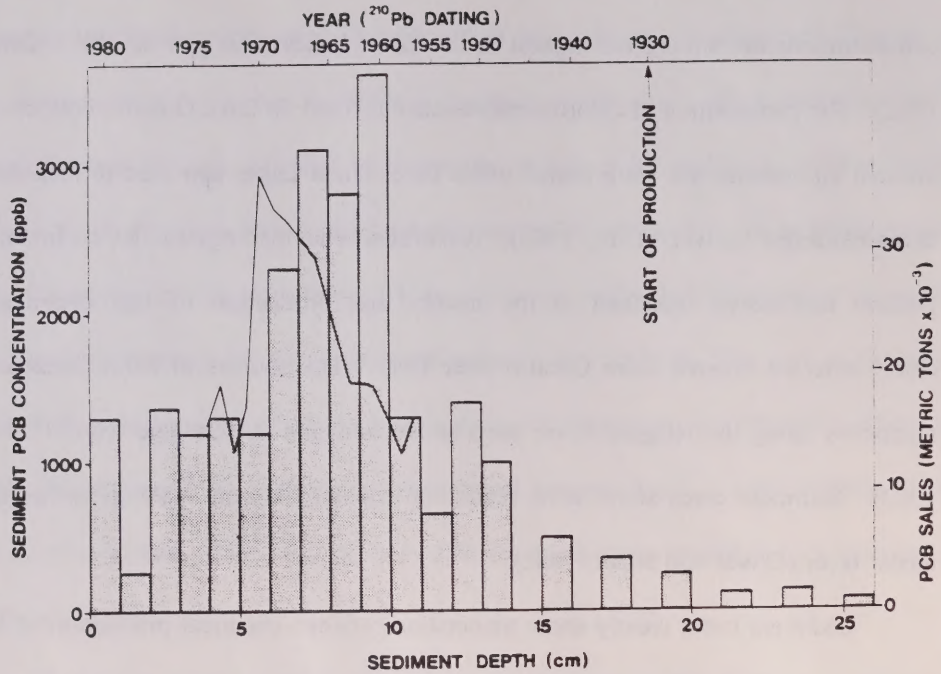

Figure 5.6.2. Total PCBs versus sediment depth in core (bottom axis) and sediment age (top axis). Sales figures for PCBs from Peakall (1975) are superimposed (right axis).

Source: Durham and Oliver (1983).

Octachlorostyrene was found in 8 of the 11 sediment cores in Lake Ontario in 1981-1982, and showed the highest concentrations in the 1960s and 1970s. Most of the Lake had concentrations of octachlorostryene less than 15 ng/g while the Niagara River concentrations were 50 -100 ng/g. The Rochester area was another hot spot with 100 ng/g of octachlorostyrene (Kaminsky et al., 1984). Concentrations of DDT, and dieldrin were two to three times higher in the depositional zones compared to the nondepositional zones. PCB

concentrations in sediments showed no significant differences among the three basins (Frank et al., 1979). Chlordane was detected in only 2% of the 229 sediment samples in Lake Ontario, and had the highest concentration of 40 ppb near Bronte Creek. Offshore concentrations declined to 5 ppb (Frank et al., 1979). Endosulfan concentrations were detected in 9% of the Lake Ontario sediment samples with a mean concentration of 2.9 ppb. The higher concentrations in the Niagara basin suggest a spill in the western end of Lake Ontario midway between the Niagara River and Toronto (Frank et al., 1979).

Metal concentrations in the surficial sediment samples from offshore Lake Ontario all exhibit maximum levels which exceed OME guidelines. The sediment concentrations of all metals except cadmium also indicate moderate to heavily polluted sediments under the EPA system. This indicates that much of the offshore sediments in Lake Ontario are contaminated with heavy metals. The sediment cores are also showing enrichment of 2 -10% of calcium carbonate over the last 80 -100 years (Mudroch, 1983).

5.6.3 BENTHIC COMMUNITY STRUCTURE

The benthic community plays an important role in the aquatic ecosystem. Benthic organisms can be an important diet item for predatory invertebrates and fishes. If the benthic community is contaminated from contact with sediments, then the benthic community can be a major route of contaminant transfer from the sediments to higher trophic levels.

Few data bases documenting benthic community structure in the offshore zone of Lake Ontario exist. Historical information on benthic communities comes from cruises between 1963-1965 from the Great Lakes Institute at the University of Toronto, the Bioindex Program

of the Great Lakes Laboratory for Fisheries and Aquatic Sciences, the International Field Year on the Great Lakes (1961, 1971), the State University of New York at Buffalo, and cruises from the Canadian Centre for Inland Waters in 1977 and 1987/1988. Most of these data are unpublished and still in raw form. Historical surveys of the benthic community have been summarized by Cook and Johnson (1974), Mozley (1990), and Nalepa (1991).

Data bases for the nearshore zone of Lake Ontario include a 1981 survey done for the Ministry of Environment, Ontario Hydro assessments around nuclear power plants, and information gathered for the RAP reports.

The species composition and abundance of the benthic community is the result of a number of factors including chemical concentrations, biological composition (fish community), and physical substrate (slope, sand, clay, depth, light). Teasing out the influence of one of these factors from the rest is a difficult and uncertain task. Therefore, fulfilling the definition of the impairment which requires a significant divergence of the benthic community structure from unimpacted control sites of similar physical and chemical characteristics will be a difficult task, given the variability in substrate type.

The Ecosystems Objectives Work Group is currently developing indicators of the benthic community for Lake Ontario, and dealing with many of these questions. The Ecosystems Working Group considers that four taxa may be considered critical for the offshore habitat of Lake Ontario: *Pontoporeia, Styoldrilius, Heterotrissocladius*, and *Pisidium conventum*. This offshore community has changed very slowly. However, there is some suggestion that *Tubifex* sp. may have replaced *Heterotrissocladius* and *Pisidium conventum*. It is more difficult to develop indicators for nearshore areas as they are highly variable

environments. Potential nearshore indicators may include: *Gammarus* (amphipods), sphaeriids, isopods, and crayfish (Ecosystem Objectives Work Group (EOWG), 1990b).

A 1981 report to the Ministry of Environment found declining levels of abundance of benthic species with increasing depth and distance offshore. *Pontoporeia hoyi* and *Styoldrilius herringianus* were dominant in offshore communities, and were replaced by tubificids near point sources of pollution. A total of 196 taxa of benthic invertebrates were recognised from the nearshore transects. Most of the differences among transects were attributable to substrate differences. Rock and sand supported the fewest taxa while coarse heterogeneous substrata generally supported the greatest number of taxa.

Invertebrate standing stocks ranged from 1,612 individuals near Port Hope to 324, 261 individuals at the Humber River. The number of taxa per station averaged 21 throughout the nearshore zones. The eastern stations and transects had higher numbers of species. The north shore generally had low abundance of invertebrates due to the exposure to wave energy and sudden temperature changes caused by thermal upwellings (Wilkins, 1985). In general, based on the distribution of species, the study area split into 5 regions: southwest, northwest, central, northeast, and east. The distribution of species normally associated with eutrophic conditions such as *S. herringianus, P.conventus, P. hoyi*, and *Hetrotrisso cladius changi* were common in the southwest and northeast. Species associated with organic enrichment such as *P. ferox, P. castertanum, B. tentaculata,* and *Procladius* sp. were found in the northwest and east. Transects around the Niagara River indicated eutrophic conditions (Integrated Exploration, 1984). Tubificids accounted for more than 50% of the total fauna at the deep stations at Niagara-on-the-Lake, Port Dalhousie, Oakville, Etobicoke, Humber Bay, Prince

Edward Point, and Amherst Island (Wilkins, 1985). Tubificid populations greater than 5000/m^2 are considered indicative of heavy organic enrichment (Wright et al., 1955 cited in Wilkins, 1985). A comparison of tubificid counts taken by the IJC in 1969 with this study indicates a greater number of stations which exceed the 5000/m^2 criteria. Eastern Ontario in particular appears to be significantly more enriched in 1981 than in 1969 (Wilkins, 1985).

Chemical contamination of the sediment was measured in 40% of the stations, and due to sampling uncertainties, no correlations among chemical contamination, substrate type, and benthic community indicators was made (Wilkins, 1985). Therefore, in one of the major studies on Lake Ontario benthos, no conclusions regarding causal pollutants can be drawn.

The Bioindex Program considered the hypolimnion of Lake Ontario to be dominated by *Pontoperia*, which comprised 87 to 90 % of the total biomass at the offshore sites in 1983. Oligochaetes were next in importance with *Stylodrilus heringianus* and *Tubifex tubifex* being the dominant species. The populations of *Pontoperia*, total oligochaetes and sphaeriid clams were all significantly lower at the offshore station than at two nearshore stations. During 1983, the total invertebrate population at the offshore station averaged 2332 individuals/m^2 compared to 7500 ind/m^2 in eastern nearshore and 5508 ind/m^2 at the mouth of the Niagara River (Johannsson et al., 1985).

In Lake Ontario, the benthic fauna in the nearshore zones is impacted around local sources of pollution, as well as at great depths off the major river mouths. Reduced numbers of oligotrophic indicator species and increased numbers of pollution tolerant species were found at 70 m off of the Niagara River and at 54 m off of the Toronto Harbour (Nalepa, 1991).

Along the southern nearshore of Lake Ontario, there is a scarcity of *Pontoporeia hoyi*, which has been attributed to chemical pollution, water temperature, or the presence of *Stylodrilus* spp.

The offshore benthic community is dominated by *Pontoporeia* followed by *Stylodrilus*, but the standing stocks of the macrobenthos are lower than might be expected given the productivity of the system (Nalepa, 1991). Although the amount of organic material settling from the epilimnion in Lake Ontario (and food for benthos) is greater than Lake Michigan or Lake Superior, benthic biomass was less than one half the biomass found in these other two lakes (Nalepa, 1991). One possible explanation for the lower standing stocks is exposure to persistent contaminants. However, no studies have been specifically designed to assess the effect of long term, low level exposures of persistent contaminants on Lake Ontario benthos. Some laboratory experiments suggest that persistent contaminants may be affecting benthos. For example, zinc concentrations between 58 and 123 $\mu g/g$ lowered the swimming activity of *Pontoporeia*, and zinc concentrations in surficial sediments of Lake Ontario range between 100-500 $\mu g/g$ (Murdroch et al., 1983).

The abundance and biomass of macrobenthos was 10,200/m^2 in 1964 compared with 5,400/m^2 in 1972. In the early 1980's, the reduced benthos in the offshore zone continued, with mean dry weight biomass of 0.83 g/m^2 compared to 2.12 g/m^2 in 1967-68 (Nalepa, 1991).

These changes in the benthic fauna of Lake Ontario from 1960's to 1970's have occurred at a time when nutrients were increasing, and the continued reduction in the 1980's, occurred during nutrient decrease. Therefore the traditional explanation of changes in benthos

being related to changes in water column productivity does not seem to explain the benthos trends. These changes in the benthos may instead be a result of chemical contamination. The decline in benthos from the 1960's to 1970's occurred at a time of peak contaminant concentrations in the sediment. However, the continued decline in benthos abundance and biomass in the 1980's is occurring at a time of reduced contaminant concentrations in the sediment.

Of the eight RAPs on Lake Ontario, four have identified degradation of benthos as an impairment: Metro Toronto, Hamilton Harbour, Niagara River (Canadian side), and the Bay of Quinte. Port Hope has a benthic community characteristic of eutrophic conditions, however it is not considered an impairment despite evidence illustrating benthic deformities and altered structure. The Port Hope RAP report states that these effects cannot be linked to chemical or radiological contamination as they cannot be separated from the influence of substrate type (Port Hope RAP Team, 1989). Tissue concentrations of both heavy metals and radionuclides were greatest at the sites with the most heavily contaminated sediment in the inner harbour (Hart et al., 1986). Degradation of benthos may exist in Oswego Harbour, based on significantly higher acute mortality of *Daphnia magna, Pimephales promelas* and *Hexagenia limbata* than controls when exposed to sediments from the harbour. *D. magna* showed no chronic effects when exposed for 21 days to sediment samples from the Harbour (New York State Department of Environmental Conservation, 1990).

Benthos in Hamilton Harbour were originally sampled in 1964 and resampled in 1984. Results from the 1964 survey supported the hypothesis that poor water quality and/or sediment toxicity was inhibiting the abundance and distribution of oligochaetes. The

community was dominated by *Limnodrillus hoffmeisteri* and *Tubifex tubifex*. In the 1984 survey, *T. tubifex* had decreased from 30% to 10% of the population and *L. hoffmeisteri* had decreased from 50 to 32%. The toxic zone around the Ottawa Street slip had decreased from 2 square km to a small area. Biomass has increased 5-20 fold and community composition has shifted from the pollution-tolerant to the pollution-sensitive species. Sphaeriids have returned and four additional chronomids were found in 1984. The abundance and composition of benthos was not correlated with sediment contamination because of the influence of low dissolved oxygen, depth, sediment grain size, and organic composition. The contaminant concentrations in the benthic macroinvertebrates was also measured. Concentrations of copper, iron, mercury, zinc, PCBs, HCB, DDE, chlordane, heptachlor, aldrin, BHC in invertebrates were 2-6 times higher in the Windemere basin and Ship Canal than the associated sediments. This indicates that the benthic community may be a route of contaminant transfer through the food chain (Hamilton Harbour RAP Team, 1990).

Deformities in benthic chironomid mouth parts have been used as an indicator of chemical contamination in sediments (Warwick et al., 1987). In Port Hope, 83% of the chironomids examined from the inner harbour had deformities in mouth parts compared to 14% in the outer harbour. A dose-response relationship was observed. A dose rate of 1 mg/g/day was thought to be sufficient to induce these deformities. However, the deformities could also be a result of the elevated levels of heavy metals in the same area or water temperature differences (Warwick et al., 1987). A similar approach was taken in Chippawa Creek, where mouth part deformities were used to determine the areas requiring dredging and to assess the effectiveness of cleanup. Deformed chironomids had significantly higher

concentrations of PAHs in their tissue than upstream samples (Dickman and Brindle, 1991). Before cleanup, 14% of the chironomids at the contaminated site had deformed "teeth" or menta. After cleanup, the frequency of abnormalities fell to 7.8% which was still elevated compared to the 3.1% rate observed in the upstream populations. Chemical analysis of the cleanup up sediments confirmed recontamination by PAHs. The ability to induce mentum deformities was established by Cushman (1984) who documented a high frequency of deformities in *Chironomus decorus* in sediments contaminated with synthetic coal tar derived oil. Cleanup effectiveness was also evaluated by benthic species density and abundance. After cleanup, a species shift occurred from coal tar-tolerant *Chironomus decorus* to coal tar sensitive *C. anthracinus*, *C. reductus*, and *Tanytarsus* spp. Chironomid density also increased from 87 individuals/m^2 to 342 ind/m^2 which was still significantly lower than the control site at 2681 ind/ m^2 (Dickman and Brindle, 1991). This study illustrates the changes in species, species abundance, and deformities which occurred in a PAH contaminated area compared to a control area upstream. PAHs are therefore considered candidates for causal pollutants.

5.6.4 SEDIMENT BIOASSAYS

Sediment bioassays are used to determine the toxicity of sediments, often as a precursor to dredging or disposal activities, recognizing that chemical analysis of sediment may not accurately predict the biological impact of the sediment (Munawar and Thomas, 1989).

A variety of sediment bioassays are used: biological testing using phytoplankton; zooplankton and fish mutagenicity testing using microtox or Salmonella/microsome assays;

and genotoxicity testing using DNA alterations. Either the sediment itself or elutriates of the sediment from a variety of extraction techniques have been used. These techniques can have the advantage of being rapid, comparatively inexpensive and standardized to test. The U.S Army Corps of Engineers and various agencies have developed standard elutriate tests which have been widely used as a predictor of the toxicity of sediments. However, recent evidence indicates that the use of elutriates of sediments may seriously underestimate the effects of dredging. For example, the bacteria level in elutriates was three times lower than in sediments and had a higher pH. These two factors can play a major role in the behaviour of contaminants (Nalewajko et al., 1989a, 1989b).

Testing of organic extracts of sediments from Hamilton Harbour using the *Salmonella*/microsome assay revealed that they contained mutagenic material and that they were capable of inducing DNA adducts in mice (Metcalfe et al., 1990). Sediment extracts from 7 of 62 stations killed 70% of *Daphnia magna* within 48 hours. The acute toxicity was correlated to the oxygen demand of the sediments. Toxicity was reduced by bubbling with oxygen for 2 days on the addition of 10-200 mg/L of lime. No simple relationship existed between sediment toxicity and elevated metal concentrations (copper, zinc, and iron) in the sediment. Sediments from deeper stations in Hamilton Harbour were also acutely toxic to *Pontoperia* within 48 hours. In contrast, sediments from Lake Ontario, Cootes Paradise and Bay of Quinte showed 15% mortality of *Pontoporeia*. Zinc, lead, iron and chromium were higher in Hamilton Harbour sediments. The sediments were not acutely lethal to *Chironomus* sp., but sublethal effects such as a significantly lower feeding rate were observed (Hamilton Harbour RAP Team, 1990).

Sediments collected from the outlet of the STP near Ashbridges Bay in Metro Torontoand from a midlake station in August 1988 were tested for toxicity using *Pontopereia hoyi*. The results showed a slightly lower rate of survival of *P. hoyi* in the sediments from Ashbridges Bay compared to the midlake control. *Hyalella azteca* had only a 40% survival in the same sediments compared to 100% survival in the Lake Ontario sediments. The percentage survival also varied with the time the sediments were collected. These results demonstrate the seasonality of sediment toxicity, and the need for caution in interpreting single sample, single time period results (Munawar et al., 1989).

5.6.5 SOURCES OF CONTAMINATED SEDIMENTS

The pattern of contamination from many organochlorines such as PCBs and octachlorosyrene in bottom sediments in Lake Ontario points to the Niagara River as a major source (Frank et al., 1979, Oliver, 1984, Kaminsky and Hites, 1984). The 1979 load of mirex from the Niagara River was estimated to be 13.3 kg, originating from historical discharges from the Hooker Chemical Corporation and continuing leakage from landfill sites (Lum et al., 1987). The only other source of mirex is the Armstrong Cork Company on the Oswego River (Holdrinet et al., 1978), which is estimated to load approximately 6 g of mirex a year to the lake (Lum et al.,1987). This historical source has contaminated a 14 km stretch in the Oswego River and will continue to provide a source of mirex to the lake. Some estimates suggest that it may be 200 to 600 years before mirex contaminated sediments are covered by "clean" sediments (Halfon, 1981). Other estimates, which include mirex transported out of Lake Ontario by migrating eels, put this figure at approximately 100 years or less (Lum et

al., 1987).

The continuous mixing of sediments from resuspension and turnover will provide a continuing source of contaminants in Lake Ontario. For example, in the nearshore areas, industrial sources can often be identified as a source by the high concentration of contaminants around an outfall. In Port Hope Harbour, the highest concentrations of metals and radionuclides were located around the outfall from a uranium refinery, Cameco.

The diffuse distribution of DDT in the offshore sediments suggests a number of small, historical sources around the lake, suggesting runoff from agricultural, urban and recreational lands and atmospheric inputs as the probable sources (Frank et al., 1979).

5.6.6 CONCLUSIONS

There is evidence of elevated concentrations of heavy metals and organic contaminants in the sediment cores from the open lake compared to precolonial times. A comparison of contamination of sediments among different locations in the lake is hampered by the lack of standard protocols for sampling, storage, and analysis. It is clear from site-specific studies that nearshore areas have significant concentrations of both metals and organics. These areas include Niagara River, Hamilton Harbour, Toronto Harbour, Port Hope, Bay of Quinte, Oswego, and Rochester. The sediment cores and surficial samples from the offshore areas in Lake Ontario also indicate elevated concentrations of contaminants. The mixing and transportation of sediments among the basins is responsible for contamination of the offshore areas, which is great distances from the point sources of pollution. The type of substrate will greatly affect the concentration of contaminants in these and in the nearshore zones. If the

definition of the impairment was based on exceedence of dredging criteria, then most of the nearshore urbanized zone and localized offshore areas would be considered impaired.

An assessment of the degradation of the benthic community in the open waters of Lake Ontario is hampered by the lack of available data. The limited data points to impairments in several of the RAP areas: Niagara, Hamilton, Toronto, and Bay of Quinte. The total number of invertebrates and abundance are reduced in the offshore areas relative to the nearshore zones, but this may be a result of physical factors such as light and substrate rather than a chemical effect. Many studies can document a change in benthic macroinvertebrate species over time, from pollution-tolerant to pollution-sensitive organisms. Few of the studies have correlated benthic community structure with contaminant concentrations. The lack of data in the offshore zone of the lake makes a judgement of the occurrence of the impairment difficult at this time.

There are few examples of sediment bioassays being performed on open lake sediment. However, given the current evidence, several areas in the nearshore zone can be considered impaired through degradation of the benthic community: Toronto, Hamilton, Bay of Quinte, Niagara River, and Oswego River. Promising developments include the work of the Ecosystems Objectives Work Group in their search to statistically distinguish benthic communities among locations, the development of new biologically based sediment criteria at the Ontario Ministry of Environment, and the use of deformities in benthic mouth parts as an indicator of impairment.

Therefore based on the results of this impairment, the following chemical should be considered as a candidate for critical pollutants for Lake Ontario: PAHs based on chironomid deformities (Dickman and Brindle, 1991) and mutagenicity under *Salmonella*/microsome test in Hamilton Harbour (Metcalfe et al., 1990).

5.7 RESTRICTIONS ON DREDGING ACTIVITIES

5.7.1 DEFINITIONS

The International Joint Commission's (IJC) guidelines for listing and delisting impariment 7, "restrictions on dredging activities", are shown below.

Listing Guideline

When contaminantion in sediments exceed standards, criteria, or guidelines such that there are restrictions on dredging or disposal activities.

Delisting Guideline

When contaminants in sediments do not exceed standards, criteria, or guidelines such that there are restrictions on dredging or disposal activities.

These listing/delisting guidelines account for differences in jurisdictional and federal standards and, in theory, emphasize both dredging and disposal activities. In practice, criteria, standards, and guidelines have been established for open-water disposal only. Dredging may still occur, regardless of how greatly the sediments are contaminated, while disposal activities are restricted. The listing/delisting guidelines focus on substances in the sediments rather than on eliminating the original sources of contamination.

Section 5.7 will focus primarily on disposal restrictions. Sediment information was also gathered from harbors, river mouths, and other areas not dredged and the contaminant data were evaluated against the relevant jurisdictional criteria to supplement the comprehensiveness of the study. Additional information on sediment contamination is discussed in Section 5.6.

Five methods of dredged spoils disposal have been used in the Great Lakes, depending on the severity of sediment quality degradation: open water, beach, reuse, upland, and confined (Sediment Work Group, 1990). Open water disposal means the transport of sediments by barge or other means to the deeper offshore waters for release, provided the quality of the sediment meets or exceeds the relevant criteria. Beach disposal is a method specific to the application of clean (chemically and physically), usually sandy, material to a nearby beach site. Reuse implies using dredged material in a suitable adjacent locality for such projects as buildup of the littoral zone, reinforcement of breakwalls and sandspits, and the buildup of shoreline areas. The upland disposal option involves moving dredged material away from the water and placing it in an approved land disposal site. Lastly, confined disposal means placing material too contaminated for disposal by other means, according to jurisdictional guidelines, in a shore-based facility designed to contain the material and to slow or stop contaminant releases into the surrounding water body. The beach and reuse disposal methods are not used often because the sediments must have rather specialized engineering properties or because they must be virtually free of contamination for either option to be viable.

The primary source of information on dredging activities in the Great Lakes is the *Dredging Register* maintained by the Sediment Work Group of the Great Lakes Water Quality Board, International Joint Commission. The register is a compendium of information on individual projects on both sides of the border including their location, volume dredged, disposal methods, and where available, the concentrations of chemical parameters in the dredged materials. Three registers documenting projects from 1975 to 1979 (Dredging

Subcommittee, 1982), 1980 to 1984 (Sediment Work Group, 1990), and 1985 to 1989 (Sediment Work Group, 1991) have been published. The data relating to projects on Lake Ontario have been extracted from these publications and are summarized in Appendix G. Because of differences in reporting procedures, the data from 1975 to 1979 will not be discussed in detail. The information in the *Dredging Register* is provided by Public Works Canada and the U.S. Army Corps of Engineers and is estimated to contain data on 95% of the activities during the 1980-89 period (Dredging Subcommittee, 1990, 1991). The remaining 5% of activities consist of small, private dredging projects.

A number of caveats are attached to the use of the chemical data reported in the *Dredging Register* (Sediment Work Group, 1990, 1991). First, sampling sites were biased towards the area to be dredged and as a result the data represent the dredged sediments only, rather than the entire harbour. Therefore, these data cannot be used to assess spatial or temporal trends in contamination at a site. Furthermore, differences in analytical methodologies and detection limits prevent interannual comparisons of sediments from the same locality. Second, the quality of the chemical data has not been independently evaluated by the International Joint Commission or any of its' boards or working groups. Chemical analyses are conducted by private laboratories in New York and Ontario following the quality assurance/quality control procedures of the reporting agencies. These procedures were not provided with the dredging project data. We have not specifically evaluated the quality of these data either, as this was beyond the scope of our study. However, the chemical data from Canadian projects are the data used in evaluating the project for open water disposal (A. Khan, Canada Department of Public Works, North York, Ontario, personal communication)

and the laboratories used in Ontario and New York did not change during the 1985-89 period.

5.7.2 EVALUATION GUIDELINES

Contaminated sediments and their management are central features of the amended Great Lakes Water Quality Agreement (GLWQA) and the commitment to restoration and rehabilitation (Great Lakes Water Quality Board, 1989b). The Parties to the GLWQA have been urged to adopt compatible guidelines and criteria for dredging activities in the Great Lakes for some time and changes are anticipated in the near future (Sediment Work Group, 1990, 1991). In response to the need for a common approach, the Sediment Subcommittee (1988a) outlined a sequential approach to assessing contaminated sediments which emphasizes biological monitoring, both *in situ* and laboratory bioassays. Much work has been done in terms of biological monitoring techniques and this is expected to be incorporated in revised guidelines. The requirement for compatibility currently applies to the disposal of dredged material, but it may be necessary in the near future to develop guidelines for the application of dredging and transportation techniques (Great Lakes Water Quality Board, 1989b). Because there is an increasing shortage of available and suitable sites for shore-based confined disposal facilities or secure upland landfill sites, there is an urgent need for the development of alternative techniques to the dredging and confinement of contaminated sediments. Currently there are no technologies to remediate large volumes of contaminated sediments *in situ* and thus the only options are either to dredge contaminated sediments or leave them in place with the expectation that they will be capped by incoming cleaner sediments (Sediment Subcommittee, 1988b). Considerable work is ongoing under Annexes

7 (Dredging) and 14 (Contaminated Sediment) of the GLWQA with respect to evaluating the environmental impacts of contaminated sediments and to developing methods to remediate the contamination, particulary bioremediation.

5.7.2.1 New York. Dredging projects in the New York waters of Lake Ontario are evaluated on a site-specific basis using guidelines developed by the U.S. Environmental Protection Agency (EPA). This approach is necessitated by the variability of sediment characteristics, the nature of the project under evaluation, the available disposal options, and local conditions. The EPA criteria (Table 5.7.1) were developed based on experience with water and sediment quality characteristics and benthic communities in harbours throughout the Great Lakes. Sediments were classified as heavily polluted if benthic organisms were absent or sharply restricted in their distributions (Dredging Subcommittee, 1982). If pollution tolerant organisms predominated in the benthic community, then the sediments were classified as moderately polluted and if benthic abundance and community composition were high, then the sediments were believed to be unpolluted or only lightly polluted.

Sediments are classified as nonpolluted, moderately polluted, or heavily polluted based on bulk sediment analysis. Each measured parameter is evaluated against the relevant concentration ranges shown in Table 5.7.1 and the overall classification of the sediments is based on the predominant classification of the individual parameters (Dredging Subcommittee, 1982). The procedure is more rigid with respect to PCBs and mercury because of their bioaccumulation potential. If the guidelines are exceeded for either PCB or mercury, then

Table 5.7.1. Guidelines for the evaluation of sediments in the Lake Ontario waters of New York. Guidelines were developed by the U.S. Environmental Protection Agency and are taken from Dredging Subcommittee (1982). Parameter values are in $\mu g/g$, dry weight, unless indicated otherwise. A dash (-) indicates that a limit was not established.

Parameter	Pollution classification		
	light	moderate	heavy
volatile solids (%)	<5	5-8	>8
COD (mg/g)	<40	40-80	>80
TKN (mg/g)	<1	1-2	>2
Total phosphorus (mg/g)	<0.42	0.42-0.65	>0.65
oil and grease (mg/g)	<1.0	1-2	>2
iron (Fe) (mg/g)	<17	17-25	>25
mercury (Hg)	<1[a]	-	-
lead (Pb)	<40	40-60	>60
zinc (Zn)	<90	90-200	>200
chromium (Cr)	<25	25-75	>75
copper (Cu)	<25	25-50	>50
arsenic (As)	<3	3-8	>8
cadmium (Cd)	-	-	>6
cyanide	<0.1	0.1-0.25	>0.25
ammonia (NH_3)	<75	75-200	>200
nickel (Ni)	<20	20-50	>50
barium (Ba)	<20	20-60	>60
PCB	<10[a]	-	-

[a] Exceeding the guideline results in a polluted classification and the sediments are unacceptable for open water disposal regardless of the other parameter measurements.

the sediments are classified as polluted and they are considered unacceptable for open-water disposal regardless of the overall classification based on the other measured parameters. In addition to the chemical measurements, other factors including elutriate test results, contaminant source(s), particle-size distribution, benthos populations, color, and odor may be

considered on a case-specific basis, although their interpretation is somewhat subjective (Dredging Subcommittee, 1982).

The EPA guidelines used in the New York waters of Lake Ontario have limited applicability as indicators of the potential hazards to aquatic ecosystems. First, because the guidelines are based on bulk sediment analysis changes in substrate type may change contaminant concentrations as contaminants are primarily associated with the finer size fractions (Dredging Subcommittee, 1982). Work in the Niagara River has demonstrated that analysis of the major and trace elements and their association with minerals in different sediment size fractions provides more relevant information than the concentrations of these elements in a bulk sediment sample (Mudroch and Duncan, 1986). Second, the individual criteria were based on informed judgement and do not attempt to relate contaminant concentrations to specific adverse impacts on aquatic biota. Third, the guidelines relate to trace metal contaminants and standard parameters and do not include the suite of organochlorine contaminants which are increasingly implicated in a variety of impairments (e.g., Sections 5.1, 5.3, 5.5). Fourth, the guidelines provide information on total concentrations only and do not necessarily represent the biologically available portion of the contaminant load under ambient conditions. Fifth, the guidelines are set for individual contaminants and do not attempt to account for synergistic interactions among chemicals within the sediments.

5.7.2.2 Ontario. Prior to 1991 dredging projects in Ontario were evaluated using the open-water disposal guidelines (OWDG) developed in the 1970s (Table 5.7.2). These guidelines were applied on a site-specific basis because of the variability in sediment charactersitics, the nature of the project being evaluated, the disposal options available, and the local conditions. Open-water disposal was allowed only when contaminant concentrations in the spoils met or was lower than the OWDG and provided that existing water uses were not affected (Persaud and Wilkins, 1976). Further, spoil material for open-water disposal had to be similar in consistency and equal to or better in terms of contaminant content, than the substrate at the disposal site. Although one parameter exceeding its guideline was theoretically sufficient to reject open water disposal as an option, in practice there was some flexibility and it was usually a combination of contaminants exceeding the relevant guidelines that was weighted heavily in the final decision.

The Ontario OWDGs were designed for the sole purpose of evaluating dredged material for open-water disposal (Persaud et al., 1991). From an operational viewpoint, the OWDGs have the same limitations as the EPA guidelines used in New York (see Section 5.7.2.1 for a discussion). Although they may be applied to other projects, the OWDGs do not address the significance of contaminants in in situ sediments. Hence, they provided minimal guidance in terms of environmental protection (Persaud et al., 1991).

The Ontario Ministry of the Environment recently proposed the Provincial Sediment Quality Guidelines (PSQGs) to replace the OWDGs (Table 5.7.2). The PSQGs are designed to protect sediment-dwelling benthos, because they are the organisms most likely to be

Table 5.7.2. Ontario's Open Water Disposal Guidelines (OWDGs) developed in the 1970s and the proposed Provincial Sediment Quality Guidelines (PSQGs) developed in 1991. All criteria are in $\mu g/g$, dry weight, unless specified otherwise. The OWDGs were taken from Persaud and Wilkins (1976) and the proposed PSQGs were taken from Persaud et al. (1991).

Parameter	OWDG	PSQG Effect level		
		none	lowest	severe[a]
volatile solids (%)	6.0	-	1	10
COD (mg/g)	50.0			
TKN (mg/g)	2.0	-	0.55	4.8
total phosphorus (mg/g)	1.0	-	0.6	2.0
oil and grease (mg/g)	1.5		1.5[b]	
mercury (Hg)	0.3	-	0.2	2
lead (Pb)	50.0	-	31	250
zinc (Zn)	100.0	-	120	820
iron (Fe)	10,000	-	2,000	4,000
chromium (Cr)	25.0	-	26	110
arsenic (As)	8.0	-	6	33
cadmium (Cd)	1.0	-	0.6	10
cyanide (HCN)	0.1		0.1[b]	
ammonia (NH_3)	100.0		100[b]	
nickel (Ni)	25.0	-	16	75
cobalt (Co)	50.0		50[b]	
silver (Ag)	0.5		0.5[b]	
manganese (Mn)	-	-	460	1,100
aldrin	-	-	0.002	8
chlordane	-	0.005	0.007	6
DDT (total)	-	-	0.007	12
dieldrin	-	0.0006	0.002	91
endrin	-	0.0005	0.003	130
heptachlor epoxide	-	-	0.005	5
mirex	-	-	0.007	130
PCB (total)	0.05	-	0.01	530
PAH (total)	-	-	(2)	(11,000)

Note: The list of organochlorine and PCB values shown here is incomplete. Guidelines were specified for specific congeners in Persaud et al. (1991). The proposed PSQGs have not been officially accepted by the Ontario Government as of December 1991.

[a] Severe effects levels for organochlorines, PCBs, and PAHs are related to $\mu g/g$ organic carbon.

[b] Parameter criterion carried over from the OWDG and are not required during routine testing, but may be requested on a case-specific basis.

directly affected by contaminated sediments (Persaud et al., 1991). This protection is primarily aimed at preventing the biomagnification of contaminants through the food chain from sediment sources (Persaud et al., 1991). The Provincial Water Quality Objectives/Provincial Water Quality Guidelines (PWQO/PWQG) provided the basis for deriving sediment values, hence the level of protection afforded by these criteria is consistent with the PWQO/PWQGs. The PSQGs are intended to provide guidance on environmental protection in relation to sediment issues ranging from the prevention of contamination to the remediation of contaminated sediments.

Three levels of toxic effects are defined by the PSQGs based on the chronic long-term effects of contaminants on benthos (Table 5.7.2; Persaud et al., 1991). The no-effect level is the highest concentration at which toxic effects have not been observed and it is the level at which biomagnification is not expected to occur. Sediments in which all parameter values are at or below the no-effect levels are not expected to have adverse chemical effects on aquatic organisms or water quality. The lowest effect level is the concentration of a contaminant that can be tolerated by most of the benthic species (Persaud et al., 1991). If a single parameter is at or exceeds the lowest effect level, then it is anticipated that the sediments may have adverse impacts on some benthic organisms. Sediments in which one or all parameters exceed the no-effect level but not the lowest effect level are not expected to have **significant** (our emphasis) effects on benthos. The severe effect level is the concentration of a contaminant that is detrimental to most benthic organisms and it is anticipated that severe disruption of the benthic community will occur. Sediments with any single parameter value at or above the severe effect level are considered highly contaminated

and likely to have significant effects on the benthic community.

Once the level of contamination has been determined, a series of tests to assess the potential biological effects are conducted so that management decisions ranging from source control to remediation can be made. Sediments in which the severe effect levels have been exceeded must be tested to determine if they are acutely toxic. The resulting course of action is based on the results of these bioassays. With respect to open-water disposal, dredged materials will be matched with the disposal area in terms of contaminants, which requires that both the material to be removed and the material in the disposal area be analyzed. A fuller discussion of the rationale, application, and decision-making process of the PSQGs is available in Persaud et al. (1991).

5.7.3 ENVIRONMENTAL IMPACTS OF DREDGING AND DISPOSAL ACTIVITIES

The environmental impacts of dredging and disposal activities will be briefly summarized in the following section to provide some background on the ecological impacts of dredging and disposal in Lake Ontario. International Working Group on the Abatement and Control of Pollution from Dredging Activities (IWGACPDA) (1975), Levings (1982), Dredging Subcommittee (1982), and Holmes (1986) should be consulted for more complete reviews of these impacts.

The environmental effects of dredging depend to a certain extent on the type of project: maintenance or capital works. Maintenance projects normally involve the removal and disposal of recent sediment accumulations at regular frequencies and locations and the dredged sediments are often contaminated with heavy metals, organochlorine compounds, oil

and grease. The environmental effects are likely to be intense but shorter-term than those from capital works projects because the sediments are usually light and shifting and seldom recolonized by benthos or aquatic plants after the initial dredging (IWGACPDA, 1975). Capital projects (new work) usually involve previously undisturbed sediments and greater volumes of material and are more physically disruptive to bottom topography (Holmes, 1986). The environmental impacts are more intense and may be longer-term than those from maintenance projects, but they are somewhat less predictable in terms of frequency and location.

The environmental impacts of dredging and disposal activities include the physical disruption of habitat, redistribution and transportation of contaminants from highly polluted to relatively less polluted disposal sites, degraded water quality, and the remobilization of nutrients and contaminants from the sediments. Many of these effects are common to both operations and may in turn alter the productivity of phytoplankton, zooplankton, benthos, aquatic macrophytes, and fish (Holmes, 1986). Changing the productivity of lower trophic levels usually leads to alterations in the fish community, often to the advantage of coarse, less sensitive, and less desirable fish species. Some of the impacts, such as degraded water quality, tend to be short-term and contained within the dredging or disposal site. Others, such as the movement of contaminants to relatively unpolluted disposal sites, are of more concern because they represent a continuing source of contaminants to aquatic organisms. Fish and other aquatic organisms bioaccumulate and bioconcentrate these contaminants and there is a large and expanding toxicological data base documenting adverse effects on growth, reproduction, and survival (see Sections 5.3, 5.4, and 5.5). The environmental impacts

associated with dredging and disposal activities are probably insignificant on the scale of the Great Lakes, but in the confined ports and harbour areas, enclosed bays, and river mouths where most dredging occurs, they are probably of great significance.

5.7.4. LAKE ONTARIO DREDGING PROJECTS

Dredging projects have been undertaken in 13 Ontario and 8 New York locations on Lake Ontario between 1975 and 1989 (Table 5.7.3). Most of the dredging activity occurs in industrialized harbours or urbanized small craft harbours, with Oshawa, Toronto, Hamilton, and Rochester being the most active dredging sites since 1980. Between 1980 and 1989 36 dredging projects removed 1,822,481 m^3 of sediments in Lake Ontario; 1,185,535 m^3 from New York and 636,946 m^3 from Ontario. The major disposal methods were open lake (71% of dredged material) and confined disposal (26%) (Table 5.7.4). In the U.S. waters of Lake Ontario only open-water disposal was used and this accounted for 65% of all the disposed sediments in Lake Ontario. In contrast, confined disposal accounted for 76% of sediments disposed in Canadian waters and open-water accounted for 17%. The majority of dredging projects (19) conducted between 1980 and 1989 were small (less than 25,000 m^3). Only 8 projects were large in that more than 100,000 m^3 were dredged.

There has been a considerable decrease in the number of projects and the quantity of material disposed since 1980. Between 1980 and 1984 1,114,180 m^3 of material was disposed from 21 projects (Sediment Work Group, 1990) compared to 708,301 m^3 of dredged material from 15 projects in the 1985-89 period (Sediment Work Group, 1991). There also has been a decline in the overall number of projects (10 in 1980-84; 2 in 1985-89) and

Table 5.7.3. Location of dredging projects and disposal methods in Lake Ontario, 1975-1989.

Location	Disposal	Year
	Ontario	
Kingston	upland	1989
Long Point	open lake	1978
Trent-Severn Waterway	confined	1980
Trenton Bay	upland	1989
Point Traverse	open lake, upland	1982, 1988
Cobourg	open lake	1976, 1977, 1978
Port Hope	open lake	1979
Oshawa	land, open lake confined	1977, 1978, 1979, 1980, 1981, 1982, 1983, 1985, 1986, 1988
Whitby	confined	1978, 1989
Toronto	confined, open lake	<1979, 1980, 1981, 1983
Port Credit	confined, open lake	1975-76, 1982
Hamilton	confined	1976, 1978, 1980, 1983, 1984
Grimsby	reused, beach	1984, 1985
	New York	
Oswego	open lake	1975, 1976, 1977, 1979, 1982
Great Sodus Bay	open lake	1975
Little Sodus Bay	beach	1975
Oak Orchard	open lake	1985
Irondequoit	open lake	1988
Rochester	open lake	1975, 1976, 1977, 1978, 1979, 1980, 1981, 1982, 1984, 1986, 1987, 1988
Olcott	open lake	1985
Wilson	open lake	1977

Table 5.7.4. Number of projects by disposal method (U.S. and Canadian figures combined), 1975-1989.

Year	Disposal method					Total
	Upland	Confined	Open	Beach	reuse	
1975	0	2	3	1	0	6
1976	0	1	5	0	0	6
1977	0	5	6	0	0	11
1978	1	3	3	0	0	7
1979	0	1	2	0	0	4
1980	0	4	2	0	0	6
1981	0	1	1	0	0	2
1982	0	1	4	0	0	5
1983	0	3	1	0	0	4
1984	0	1	2	0	1	4
1985	0	1	2	1	0	4
1986	0	0	2	0	0	2
1987	0	0	2	0	0	2
1988	1	0	3	0	0	4
1989	2	1	0	0	0	3
Total	4	24	39	2	1	70

quantity of material (466,193 m^3 in 1980-84; 18, 693 m^3 in 1985-89) disposed in confined sites over the past decade. High cost, difficulties in disposal of contaminated sediments, and limited space are probable reasons for the decrease (Sediment Work Group, 1991). Fewer confined disposal facilities are being constructed and a continuing decrease in the use of confined disposal throughout the Great Lakes is anticipated (Sediment Work Group, 1990).

Dredged material disposal activities have not been restricted in the New York waters of Lake Ontario because the predominant classification of the individual parameters was non-polluted. For purposes of the analysis which follows, parameter values indicative of moderate

or heavy pollution will be treated as if they are sufficient to restrict dredging activities. All of the projects in Rochester Harbor and the majority of projects in Oswego Harbor exceeded two or more contaminant criteria in the sediments that were dredged.

The individual parameters exceeding their non-polluted concentration ranges in New York waters is limited (Table 5.7.5). The list includes COD, nutrients such as TKN and total phosphorus, and metals such as copper, zinc, and arsenic. Oil and grease and nickel also occasionally exceeded their non-polluted guidelines. Total phosphorus and arsenic were most frequently indicative of heavy pollution. However, the majority of cases occurred in the 1975-1979 period when reporting procedures were different than the procedures currently used. Since 1980 total phosphorus levels in Rochester Harbor have consistently been classified as heavily polluted and there is one project in Olcott in which lead, cadmium and zinc were classified as heavily polluted.

Dredging activities in Ontario waters during the 1975-1989 period were guided by the OWDGs. In the analysis which follows, the contaminant parameter values were evaluated against the proposed PSQGs because they are consistent with the objective of this study of indentifying causal pollutants based on cause-effect relationship. The results of this analysis should not differ greatly from those produced using the OWDGs since the lowest effect levels for metals and conventional pollutants in the PSQGs are similar to the older OWDG values. Furthermore, dredged sediments were not analyzed for organic chemicals, with the exception of PCBs, because guidelines for these substances were not previously established.

Table 5.7.5. New York dredging projects in which individual contaminant classifications were indicative of moderate or heavy pollution, 1975-1989. Data were taken from Dredging Subcommittee (1982) and Sediment Work Group (1990, 1991). Projects for which all parameters were classified as non-polluted are not shown. All contaminant concentration data are shown in Appendix G.

Location	Year	No. of parameters measured	Contaminant classification	
			moderate	heavy
Oswego Harbor	1975	14	COD, TKN, Pb, Cu, Zn	TP, As
Oswego Harbor	1976	14	COD, TKN, Cu, Zn	TP, As
Oswego Harbor	1977	14	COD, TKN, Cu, Zn	TP, As
Oswego Harbor	1979	14	COD, TP, Cu, Zn	As
Oswego Harbor	1982	14	TP	
Oswego Harbor	1984	14	TP	
Great Sodus Bay	1975	6	oil and grease	TKN, TP
Oak Orchard Creek	1985	19	TP, As	
Rochester Harbor	1975	14	TP, CU, Zn, Ni	As
Rochester Harbor	1976	14	TP, Zn, Cr	As
Rochester Harbor	1977	14	TKN, Cu, Zn, Ni	TP, As
Rochester Harbor	1978	14	TP, Cu, Zn, Ni	As
Rochester Harbor	1979	14	TP, As, Cu, Zn, Ni	
Rochester Harbor	1980	13	COD, TKN, As, Zn	TP
Rochester Harbor	1981	13	COD, TKN, As, Zn (based on 1980 data)	TP
Rochester Harbor	1982	13	COD, TKN, As, Zn (based on 1980 data)	TP
Rochester Harbor	1984	13	COD, TKN, As, Zn (based on 1980 data)	TP
Rochester Harbor	1986	14	TP, As, Cu (based on 1985 data)	
Rochester Harbor	1987	14	TP, As, Cu (based on 1985 data)	
Rochester Harbor	1988	14	TP, As, Cu (based on 1985 data)	
Olcott	1985	19	TP, As, Cr, Ni	Pb, Cd, Zn
Wilson	1977	7	COD, TKN	

TP - total phosphorus; COD - chemical oxygen demand; TKN - total kjeldahl nitrogen.

The most frequently exceeded PSQG in Ontario dredging projects between 1975 and 1989 was the lowest effect level. The contaminants listed in Table 5.7.6 are grouped under the highest PSQG level that each exceeds. Volatile solids, total phosphorus, TKN, cadmium, nickel, and lead were the parameters most frequently (>10 projects) exceeding their lowest effect guidelines (Table 5.7.6). Oil and grease, copper, chromium, and PCBs exceeded the lowest effect level less often (<10 projects) and zinc, mercury, and arsenic occasionally exceeded lowest effect levels. Because no-effect levels have not been established for most of the contaminants previously evaluated in dredged sediments, only PCBs are listed as occasionally exceeding the no-effect level. Volatile solids, total phosphorus, zinc, chromium, and nickel were the substances most often exceeding their severe effect levels, though not on a consistent basis.

Based on the above assessment for projects between 1980 and 1989 volatile solids, total phosphorus, total kjeldahl nitrogen, cadmium, chromium, nickel, and lead appear to be the pollutants most often causing restrictions on dredging activities. Oil and grease, copper, zinc, mercury, arsenic, and PCBs were less likely to lead to restrictions, but can be added to the list of causal pollutants. Independent sediment analyses from Lake Ontario generally support this assessment and suggest also that iron be added to the list of pollutants restricting or potentially restricting dredging activities (Table 5.7.7). Furthermore, these analyses indicate that Hamilton Harbour has the most severely contaminated sediments in Lake Ontario. Although some parameter guidelines were technically exceeded in New York, dredging activities were not restricted, but are treated as if they were for this analysis.

Table 5.7.6. Ontario dredging projects effects level classifications of individual contaminants, 1975-1989. Data were taken from Dredging Subcommittee (1982) and Sediment Work Group (1990, 1991). Contaminants with concentrations below the lowest level for which a guideline has been established are not shown. All contaminant concentration data are shown in Appendix G.

Location	Year	No. of parameters measured	PSQG effect level exceeded		
			none	lowest	severe
Kingston Harbor	1989	14		Cd, TP, TKN, Cu, Zn Cr, Ni	VS
Trent-Severn Waterway	1980	6		oil & grease, TP	VS
Trenton Bay	1989	13		VS, TP, TKN, Cd, Cr, Ni	
Point Traverse	1982	9		TKN, PCB, Cd	
Point Traverse	1988	14	PCB	VS, TKN, Ni	
Cobourg	1976	6		VS, TP, TKN	
Cobourg	1977	6		VS, TP, TKN	
Cobourg	1978	6		VS, TP, TKN	
Port Hope	1979	4		VS, TP	
Oshawa	1977	6		VS, TKN	
Oshawa	1978	6		VS, TKN	
Oshawa	1979	13		VS, oil & grease, TP TKN, Pb, Zn, PCB	Cr, Ni
Oshawa	1980	12	PCB	VS, Pb, Cu, Ni	Cr
Oshawa	1982	12		TKN, PCB, Pb, Cd, Cr Ni	
Oshawa	1983	12		VS, TP, TKN, PCB, Pb Cd, Cr, Ni	
Oshawa	1985	13		VS, TKN, TP, Pb, Zn Cr, Ni	PCB
Oshawa	1986	13	PCB	VS, Cd	
Oshawa	1988	14		VS	
Whitby	1978	13		oil & grease, TKN, TP PCB, Pb, Cu, Zn	VS Ni, Cr
Whitby	1989	14	PCB	VS, Hg, Cd, Cu, Cr, Ni	
Toronto	<1979	11		VS, TKN, oil & grease TP, PCB, Pb, Zn, Ni Cu, Cr, Hg	
Toronto	1980	11		VS, TKN, oil & grease TP, PCB, Pb, Zn, Ni Cu, Cr, Hg	

Table 5.7.6. Continued.

Location	Year	No. of parameters measured	PSQG effect level exceeded		
			none	lowest	severe
Toronto	1981	11		VS, TKN, oil & grease TP, PCB, Pb, Zn, Ni Cu, Cr, Hg	
Toronto	1983	6	PCB		
Port Credit	1975-76	5		VS, oil & grease, TKN	
Port Credit	1975-76?	8		VS, oil & grease, TKN, Pb	
Port Credit	1982	9		TKN, Cd	
Hamilton Harbor	1983	14		VS, oil & grease, TKN	TP, Zn Cr
Hamilton Harbor	1983	14		VS, oil & grease, PCB Hg, Pb, As, Cd, Cu, Ni	TP, Zn TKN,Cr
Hamilton Harbor	1984	11	PCB	TP, Cd, Ni	
Grimsby	1984	8	PCB	Pb, Cu, Cr, Ni	

VS - volatile solids; TP - total phosphorus; TKN - total kjeldahl nitrogen; COD - chemical oxygen demand.

Some of the disposal restrictions are specific to U.S. or Canadian waters. For example, PCBs were found to exceed Ontario guidelines in 10 projects whereas in New York this did not occur, perhaps because the criterion used in New York is three orders of magnitude greater than the no-effect level in Ontario. Arsenic was the most frequently observed pollutant exceeding the non-polluted category in New York whereas in Ontario only two projects reported high arsenic concentrations. The Ontario arsenic no-effect criterion is double that used in New York.

Table 5.7.7. Pollutants exceeding open water disposal guidelines from Lake Ontario harbours reported independently.

Location	PSQG effect level exceeded			Reference
	none	lowest	severe	
Port Hope				
turning basin		Zn, Cd, Mn, Co, As, Cr Ni, PCB	LOI, Fe, Cu Pb	1
outer harbour		Cd, As, PCB	Fe	1
Ganaraska River		Zn, Cd, Mn, Cu, As, Pb Cr, Ni, PCB	LOI, Fe	1
Western Lake Ontario				
off Niagara River[a]		Pb, Zn, Cr, Hg	Ni, Cu	2
Hamilton Harbour[a]			Fe, Mn, Cr Zn, Pb, Cu Ni, Cd	3
Toronto Harbour[a]		Fe, Mn, Zn, Pb, Cu, Ni Cd	Cr	3

References: 1 - Hart et al., (1986); 2 - Mudroch (1983); 3 - Nriagu et al., (1983).

[a] Surficial sediments down to a depth of 10 cm.

Some of the differences in causal pollutants may be attributable to different analytical protocols, in that not all contaminants in the guidelines were investigated at a given site. This would confound the above analysis for causal pollutants because it is not obvious in the available databases if this occurs at a particular location. Some of the differences also may be related to industrial activities with different waste streams, which are subsequently reflected by sediment contamination. For example, steel manufacturing is the dominant

industrial activity in Hamilton Harbour whereas manufacturing photographic films is the predominant industry in Rochester.

All of the Remedial Action Plans (RAPs) for Lake Ontario Areas of Concern (AOCs), except Oswego, list restrictions on dredging activities as an impaired use. Dredging in the Bay of Quinte is restricted because some heavy metals exceed the disposal guidelines; current conditions are being evaluated (Bay of Quinte RAP Team, 1990). Sediments in the turning basin and west slip of Port Hope Harbour contain elevated levels of heavy metals, PCBs, and radionuclides (Port Hope RAP Team, 1989). Because of the radionuclides, dredged material would have be be disposed in a low-level radioactive waste management facility which is not presently available. A new harbour was created at Port Hope because of the disposal problems and because of continuing sedimentation. Sediments in most embayments in the Metro Toronto region exceed the disposal guidelines for a number of parameters and have been subjected to Environmental Assessment proceedings in the past and are likely to continue to be in the future (Metro Toronto RAP Team, 1988). Dredging has occurred in the Keating Channel for navigational and flood protection purposes and the sediments were deposited in the Leslie St. spit. Hamilton Harbour sediments grossly exceed acceptable limits for open water disposal and must be placed in a confined disposal facility whose capacity is only adequate until 2000 (Hamilton Harbour RAP Team, 1989). There are currently no restrictions on disposal of dredged sediments from Oswego Harbour, but there is concern about open-lake disposal because cyanide, zinc, lead, barium, and oil and grease exceed the EPA's nonpolluted guideline at some sampling sites in the harbour (New York State Department of Environmental Conservation, 1990).

5.7.5 CONCLUSIONS

Volatile solids, total phosphorus, total kjeldahl nitrogen, cadmium, nickel and lead are the causal pollutants most frequently restricting dredging activities in Ontario projects in Lake Ontario. Oil and grease, copper, chromium, and PCBs were causal pollutants as well, but less often. Arsenic, zinc, copper, total phosphorus, COD, and total kjeldahl nitrogen would be the pollutants most likely to restrict dredging activities in New York waters. Although in some New York projects various contaminant concentrations are indicative of moderately to heavily polluted sediments, using the EPA guidelines all dredgate was disposed in open waters.

The Ontario and New York dredging data were adequate for determining which of the conventional inorganic pollutants (nutrients, heavy metals) were causal pollutants leading to restrictions on dredging activities. Until recently however, guidelines for organic chemicals had not been promulgated, with the exception of the PCB guidelines in both jurisdictions. Because there were no guidelines for organic compounds prior to 1991, sediments dredged between 1975 and 1989 were not analyzed for these substances. Hence our analysis of the data and the conclusions we made concerning causal pollutants are unavoidably biased. As a result of the development of the PSQGs, Ontario will require routine assessment of the conventional inorganic chemicals and organic chemicals with respect to future sediment issues, including dredging and dredgeate disposal (Persaud et al., 1991).

5.8 EUTROPHICATION OR UNDESIRABLE ALGAE

5.8.1 DEFINITION

The International Joint Commission's guideline for use impairment 8, "eutrophication or undesirable algae" as shown below.

Listing Guideline

When there are persistent water quality problems (e.g., dissolved oxygen depletion of bottom waters, nuisance algal blooms or accumulation, decreased water clarity etc.) attributed to cultural eutrophication.

The listing guideline does not provide a definition for "persistent", "nuisance", or "decreased". Unlike other impairment guidelines, this definition does not specify that exceedence of guidelines for phosphorus, nitrogen or dissolved oxygen constitutes grounds for considering the use impaired. The rationale for these guidelines is given as consistency with Annex 3 of the Agreement, and accounting for persistent problems (IJC, 1991). An area may be delisted with respect to eutrophication or undesirable algae as shown below.

Delisting Guideline

When there are no persistent water quality problems (e.g. dissolved oxygen depletion of bottom waters, nuisance algal blooms or accumulation, decreased water clarity etc.) attributed to cultural eutrophication.

Eutrophication is the process of change in the quality of a water body caused by increased plant nutrients (NRC/RSC, 1985). When this condition results from nutrients of human origin, it is called "cultural eutrophication". Many limnologists believe that phosphorus (P) is the limiting factor in lakes and that the addition of P causes increases in algal productivity and various secondary consequences such as de-oxygenation and loss of

fish species (Likens, 1972; Schindler, 1977). Orthophosphate has been considered the limiting nutrient for algal growth in most of the Great Lakes (Beeton, 1969). Other nutrients, such as nitrogen may become limiting when P concentrations are high. Silica can also be limiting for diatom growth (Welch, 1978).

Information on eutrophication in Lake Ontario comes from three main sources: cruises on Lake Ontario taken by Environment Canada; nearshore information from RAP reports, Ontario Ministry of Environment (OME) and New York State Department of Environmental Conservation data (NYDEC); and academic workers. This discussion will review the available evidence on phosphorus concentrations and loads, nitrogen concentration and loads, dissolved oxygen concentrations, algal biomass, and water clarity in Lake Ontario. Changes in phytoplankton community structure will be discussed under impairment 13: degradation of phytoplankton and zooplankton populations. Studies are summarized in Appendix H.

5.8.2 PHOSPHORUS CONCENTRATIONS AND LOADINGS

5.8.2.1. Historical overview. The first comprehensive study on the trophic state of Lake Ontario was completed in 1916, and documented the bacterial problem from raw sewage discharges. Increasing reports of a change in the trophic state of Lake Ontario began in the late 1950's and 1960's. The mean annual biomass of phytoplankton at the Toronto Island water intake showed a doubling from 1923 to 1954 and a change in the community composition (Schenk and Thompson, 1965, cited in Stevens and Neilson, 1987). In 1964, the governments of Canada and the United States directed the IJC to study and report on the occurrence, extent, sources, and cause of transboundary pollution and possible remedial

measures. For the first time, a comprehensive assessment was made of the nutrient status of Lake Ontario. The report stated that Lake Ontario was morphometrically oligotrophic, but had water quality typical of mesotrophic conditions, with localized areas of eutrophic conditions. Phosphates from detergents were estimated to contribute 70% of the total phosphorus load, and the report recommended a reduction in phosphorus levels in detergents and improved phosphorus removal from sewage treatment plants. They also estimated that admissible loadings to Lake Ontario were 0.4 $g/m^2/yr$ and dangerous loadings were 0.8 $g/m^2/yr$. The estimates of phosphorus load at that time were 0.7 $g/m^2/yr$ with projections of 1.4 $g/m^2/yr$ by 1986.

In response to these findings, limits of 20% P_2O_5 or 8.7 % P by weight were placed on Canadian detergents in 1970 and reduced to 5% P_2O_5 or 2.2% P by weight by 1972. This P limit in detergents is still the current Canadian regulated limit, now under the Canadian Environmental Protection Act. New York and Indiana followed with limits of 20% $P_2$05 in 1970 and 1.1% P_2O_5 in 1972. The current U.S. limit in Great Lakes States is 0.5% P by weight, or one-quarter of the Canadian limit. The Canadian limits apply to all detergents, whereas some of the U.S limits only apply to household detergents. Some states such as Minnesota (25% P_2O_5) and Wisconsin (19.75% P_2O_5) have moved to regulate P in dishwasher detergents (Leonardelli, 1990). The Great Lakes Water Quality Board (GLWQB) (1989b) recommended that Canada implement legislation to reduce the P content in laundry detergents to 0.5% P by weight.

The 1972 Great Lakes Water Quality Agreement set a limit of 1.0 mg/L phosphorus on the effluent from sewage treatment plants discharging more than 1 million gallons a day.

A major study, The Pollution From Land Use Activities Reference Group (PLUARG), grew out of the 1972 agreement and reported on a broad range of measures required to reduce P and toxic loadings to the Great Lakes. This study helped form the basis for the 1978 Agreement. Annex 3 of the 1978 Great Lakes Water Quality Agreement dealt specifically with P control measures which were to "... minimise eutrophication problems and prevent degradation with regard to phosphorus in the boundary waters of the Great Lakes basin." The specific objective for Lake Ontario was the" ... reduction in present levels of algal biomass to below that of a nuisance condition in Lake Ontario including the International Section of the St. Lawrence River and the elimination of algal nuisance [growths] in bays and in other areas wherever they occur".

The 1978 agreement also set the 1976 load of P in Lake Ontario as 11,000 tonnes/year with a future target P load of 7000 tonnes/year. This target future P load was estimated to be equivalent to 10 μg P/L, chlorophyll a level of 2.6 μg/L and an average Secchi depth of 5.3 m in the open waters of Lake Ontario (Stevens, 1988). A deadline of 1980 was set to confirm this future load of 7000 tonnes for Lake Ontario, to allocate loads, and to develop compliance schedules. The 1978 agreement specified the maximum P discharged from STPs over 1 million gallons per day to be 0.5 mg/L in the Lower Great Lakes.

In 1983, three years after the deadline, these P allocations and compliance schedules were established for most areas and recorded as a supplement to Annex 3. The 0.5 mg/L effluent limit for STPs over 1 million gallons was not confirmed, and the limit was doubled to 1 mg/L, the same limit for the Upper Lakes. The target load for Lake Ontario was confirmed at 7000 tonnes/year, which required a further reduction of 1,210 tonnes/year.

However, the P load allocations and compliance schedules were still not specified for Lake Ontario, but were promised within one year. The 1987 Agreement revised this load reduction from 1,210 to 430 tonnes/yr (GLWQB, 1989a). The Canadian position was that Canada's share of the 430 tonne reduction was within the noise of the model and Lake Ontario should be given time to respond to the 1 mg/L limit, so no new programs were required. The United States position was that P reduction was still required, and they took 235 tonnes as their share of the Lake Ontario reduction, which implies a 195 tonne/year reduction for Canada. There are still no load allocations or compliance schedules for the Canadian side of Lake Ontario, and progress on P reductions from New York is behind schedule (GLWQB, 1989a). The most recent U.S. report to the IJC claims a load reduction of 106 of the 235 tonnes/year required. However, no documentation was given to support this reduction. Due to problems in reporting, the IJC could not estimate P loads in 1988.

In 1969 total phosphorus loads to Lake Ontario were estimated to be 14,600 tonnes/yr, the total declined to 6,680 in 1983 and increased to 8,000 in 1984 and 9,500 tonnes in 1986 (Stevens and Neilson, 1987; Stevens, 1988; Water Quality Board, 1989). According to the Great Lakes Water Quality Board (1985), the total loading objectives had been approximately achieved in both Lake Erie and Lake Ontario in 1981. However, the achievement of the P target load in Lake Ontario should be viewed with caution. There are difficulties in the calculations surrounding the P loadings. There is no indication of the reliability of this loading estimate, and widely different results were found during a laboratory roundrobin; some laboratory results may be biased (GLWQB, 1989a). Many tributaries are not monitored for P concentrations. In the monitored streams, the frequency of monitoring

can make a large difference in the results obtained. When the frequency of sampling was increased from monthly to approximately weekly, the estimates of P loads changed ranged from -30 to +35% with the estimates of total phosphorus loading from the Humber River in Ontario increasing by 98% (Stevens, 1988). The natural variations in river flow will also result in variations in the P load to Lake Ontario. The P load from U.S. tributaries can range from 800-1400 tonnes/yr or 11-20% variation.

The contribution of P from municipal STPs changes depending on whether actual flows or design flows are used in the calculations. Using design flows as a basis of calculation, increases P loadings from 1,717 tonnes/year to 2,132 tonnes/yr or 20% increase. The total P loads to Lake Ontario do not take into account how much of the load is assimilated or retained in nearshore environments. Total P loads into the Bay of Quinte, Hamilton Harbour and Toronto Harbour, will have less of an effect on the main lake than if loaded directly into the lake. For example, retention of P in Hamilton Harbour was 66%, resulting in 60 tonnes/yr being exported to the lake (Stevens, 1988).

The increasing population around Lake Ontario has the potential to eliminate the gains made in P reductions over the past 20 years (Leonardelli, 1990). There are plans to expand a number of STP in the Lake Ontario basin in the next 10 years to serve an increased population, which will cause increased P loadings to Lake Ontario. For example, the Main STP in Toronto is scheduled to double in capacity. The impact of increasing population and hence P loads, may make it necessary to further decrease the P concentrations in the effluent of STPs to the planned 0.5 mg/l, to further reduce the P content in detergents and all products and to limit nonpoint runoff. The Great Lakes Water Quality Board has calculated that

additional P reduction in detergents would reduce P loadings to the Great Lakes by 6%.

5.8.2.2. Phosphorus concentrations. Phosphorus concentrations in the offshore waters of Lake Ontario are measured during annual cruises by Environment Canada. This data set contains information on both the spring and summer total phosphorus (TP) concentrations at a variety of stations. These concentrations can be used as an indicator of the trophic state of the lake, and hence the degree of impairment from eutrophication.

Average spring TP concentrations in Lake Ontario have decreased significantly since the 1960's. Spring concentrations have decreased approximately 1 μg/L annually since 1969 (Stevens and Neilson, 1987). In 1968, the spring P concentration was 25 μg/L and increased to 30.6 μg/L in 1973 compared to 17 μg/L in 1979 (Kwiatkowski and Neilson, 1983) (Stevens and Neilson, 1987) and in 1981 was 12.5 -15 μg/L and in 1982 was 11.0 - 13.0 μg/L (Stevens, 1988). In 1985 spring surface midlake TP concentrations averaged 10.7 μg/L, compared to the target level of 10 μg/L. Spring TP concentration has remained close to 10 μg/L since 1985 (GLWQB, 1989a). Concentrations of TP in the Niagara River show a similar decrease of approximately 1 μg/L annually from 1976 to 1983 (Kuntz, 1988).

Localized inputs of TP were noted in the spring from Toronto, Niagara, Genesee, Oswego, Black River, Hamilton, and the Bay of Quinte. In the fall nearshore regions ranged from 25- 30 μg/L compared to 10-12 μg/L midlake. The soluble nutrients such as soluble reactive phosphorus silica, total filtered phosphorus and filtered nitrate-plus-nitrite were inversely correlated with phytoplankton indicating the influence of phytoplankton growth cycle (Neilson and Stevens, 1987). In contrast to earlier model predictions, Stevens and Neilson (1987) stated that 80% of the spring TP could be accounted for by loadings from the

previous two years and 90% from the previous three and four years.

Soluble reactive phosphorus (SRP) concentrations in Lake Ontario were significantly correlated to TP, indicating that the form of P likely to be biologically available was also decreasing significantly (Stevens and Neilson, 1987). Spring surface SRP showed a significant decrease at a rate of 0.83 μg/L from 1973 to 1985 (Stevens, 1988). Spring concentrations of SRP were 7.3 μg/L in 1968 and 6.2 μg/L in 1979 (Kwiatkowski and Neilson, 1983). Since 1985, SRP concentrations are generally less than 5 μg/L (GLWQB, 1989a). SRP concentrations are fairly constant throughout the lake, and lowest in the Kingston Basin (Stevens, 1988). However, concentrations of TP during the summer in the epilimnion did not show this steady decrease over time. In 1969, the concentrations of TP in the summer were 16.7 μg/L and in 1982, the concentrations were 14.4 μg/L (Stevens and Neilson, 1987). Since the 1980's, mean summer TP may exceed spring TP by 1-2 μg/L.

The OME guideline to protect aquatic life is 20 μg P/L. Above this level, nuisance algal blooms can occur. Concentrations of P above this 20 μg/L guideline do occur in localised areas in Lake Ontario including Toronto, Hamilton, Port Hope, Bay of Quinte, Welland River, Rochester, Black River, and Oswego (Stevens, 1988).

A Composite Trophic Index (CTI) was developed during PLUARG activities to indicate the trophic status of a lake. The zone of Lake Ontario around Toronto, Hamilton and Niagara, has the highest average concentrations of nitrate-plus-nitrite and ammonia and organic nitrogen, with a CTI ranging from 5.4 to 8.6 which indicates mesotrophic conditions (Stevens, 1988).

5.8.3 NITROGEN CONCENTRATIONS AND LOADS

Unlike TP concentrations, nitrogen concentrations expressed as nitrate have shown a steady, significant increase in Lake Ontario. Spring nitrate concentrations have increased from 230 to 375 μg n/L from 1968 to 1982 (Stevens and Neilson, 1987). Since 1969, nitrate and nitrite concentrations have increased at the rate of 9.5 μg/L/yr, which translates into an increase of 15,510 tonnes/year (Stevens, 1988). This increase in nitrogen concentrations is also observed in the Niagara River (Kuntz, 1988). Nitrate loads from the Niagara River to Lake Ontario have increased approximately 40% since 1975, from approximately 3,140 tonnes/yr in 1975 to 5,100 tonnes/year in 1983 (Kuntz, 1988). Nitrate concentrations are increasing in Lake Huron, Erie, and Superior, suggesting that a common source such as atmospheric deposition may be responsible. However, dissolved organic nitrogen has shown no significant trend from 1972.

An important indicator of trophic status is the nitrogen-phosphorus ratio. This ratio has been used as a gross predictor of blue-green algal blooms. Excess concentrations of nitrate and nitrite in surface waters in summer may prevent the formation of nitrogen fixing blue-green algal blooms. Increases in the total N:total P ratio are associated with a shift from blue-greens to green algae and diatoms. With increases in nitrate concentrations and decreases in TP concentrations, the N:P ratio has increased greatly since 1972. Spring N:P ratios from 1969 to 1974 were approximately 10-12, and increased three fold to 30 in 1982 and 40 in 1987 (Stevens, 1988; GLWQB, 1989b).

Unlike the trend of increasing nitrate and nitrite concentrations in Lake Ontario, ammonia concentrations have shown a steady decline from 1969 to 1980. Concentrations are now below 5 μg N/L (Dobson, 1984). Offshore concentrations of ammonia show little variability and many areas are below 1 μg/L (Stevens, 1988). Elevated concentrations are seen in the nearshore area, especially around Toronto, Bay of Quinte, Black River, and the southern shore from the Niagara to the Genesee River, ranging from 11-92 μg/L in the spring (Stevens, 1988). Concentrations often increased in the summer as ammonia from fertilizers undergoes nitrification in the soil in warm weather. Agricultural runoff will therefore contribute ammonia in the spring and nitrate in the summer.

Nitrogen concentrations are not subject to any management plan, because algal growth was considered to be P limited and because of the difficulty in controlling N sources (Stevens, 1988).

5.8.4 DISSOLVED OXYGEN CONCENTRATIONS

The Great Lakes Water Quality Agreement Objective for dissolved oxygen in the connecting channels and in the upper waters of the Lakes is 6.0 mg/L at any time, and in the hypolimnion, dissolved oxygen "... should not be less than necessary for the support of fishlife, particularly the cold water species". Oxygen solubility decreases in a curvi-linear manner with increasing temperature. Lack of dissolved oxygen (DO) can be a problem in the hypoliminion as stratification of the lake continues. Except in localized areas, DO is not generally biologically limiting in Lake Ontario, and remains close to 100% saturation for most of the year at stations at depths of 50 m and greater (Sly, 1991).

In the nearshore zone, the concentrations of DO can be highly variable. The vertical distribution of dissolved oxygen concentrations has been used to identify the trophic status of a lake. Oligotrophic lakes typically have a pattern of increasing DO concentrations below the thermocline. Eutrophic lakes typically have a pattern of decreasing dissolved oxygen concentration below the thermocline (Wetzel, 1975). Low levels of DO have been documented in polluted nearshore areas such as Hamilton Harbour (Hamilton Harbour RAP Team, 1989), and not found to be a problem in other areas such as Niagara River, Rochester and Oswego (Bertram, 1985). Two - thirds of the stations sampled at Rochester and almost all stations at Oswego showed trends of decreasing DO with depth, indicating a eutrophic pattern.

Oxygen in the offshore, near-surface waters of Lake Ontario follow a seasonal pattern, with maximum values from January to June of 13 to 15 mg/L and minimum values in July/August/September ranging from 9- 11 mg/L (Dobson, 1984). In the deep water (colder than 4°C) of Lake Ontario, oxygen was depleted at a rate of 0.3 mg/L every month. Dissolved oxygen in the Kingston basin can become reduced when there is a lack of wind driven circulation (Sly, 1991).

5.8.5 ALGAL BIOMASS

One of the most visible manifestations of eutrophication is the growth of *Cladophora*, a green filamentous algae which attaches to rocky shorelines in most of the Lower Great Lakes. *Cladophora* growth is controlled by light, temperature, nutrients and availability of substrate. The algae can store phosphorus, and so does not always respond in a predictable

way to decreasing P concentrations in the lake. Concentrations of internal P are significantly correlated to the biomass of *Cladophora* (Millner and Sweeney, 1982). The average level of internal P in *Cladophora* was 0.49% in 1972 in Lake Ontario, and decreased to 0.26% in 1982 and 0.29% in 1983. The biomass and growth rate of *Cladophora* in 1982 was less than half of those in 1972 (Painter and Lamaitis, 1987). This decrease was attributed to the P control programs in the early 1970's (Painter and Kamaitis, 1985). Several reports indicate that internal P concentrations in *Cladophora* are reaching limiting levels (Painter and Kamaitis, 1985). This is expected to lead to further reductions in the growth of *Cladophora* in Lake Ontario.

In the spring, with increasing water temperature and increased light, *Cladophora* grows in length rapidly (1.5 cm/day) (Bellis and McLarty, 1967). *Cladophora* has a distinct two-phase growth cycle with periods of intensive growth in May-June and again in late August-September. From late June to August, filaments of *Cladophora* can easily become detached (sloughed off) to form large floating or submerged mats (Neil and Jackson, 1982). Mats of *Cladophora* can become a public nuisance due to fouling of fishermen's nets, clogging of intake pipes, piling up on beaches, and the production of rotten odors. Rotting *Cladophora* on beaches became the symbol of the eutrophication of the Lower Great Lakes, and remains a public concern in localized areas.

Remote sensing has been used to determine the geographic extent of *Cladophora*. In the 1970's in the western portion of Lake Ontario, approximately 66% of the nearshore standing zone was covered with *Cladophora* (Wezernak et al., cited in Neil and Jackson, 1982). Reductions in the standing crop of *Cladophora* of 50-60 % have been observed

between 1972 and 1982-1983 on Lake Ontario (Stevens, 1988).

An additional response to phosphorus is the growth of phytoplankton. This can be measured in a variety of ways, including chlorophyll a, biomass (mg/L), Secchi disc, particulate organic carbon, and particulate organic nitrogen. These methods are not directly comparable. Spring concentrations of chlorophyll a were 5.5 μg/L in 1974 and had decreased to 3.3 μg/L in 1979 (Kwiatkowski and Neilson, 1983). There was no significant trend in chlorophyll a measurements taken in August/ September from 1967 -1981 (Dobson, 1984). However, changes in the methodology of chlorophyll a collection by at Environment Canada make time trend analysis difficult. Before 1972, 1 m samples were taken, in 1972, a number of depths were sampled, and after 1974 integrated 0-20 m samples were taken.

Phytoplankton biomass in the nearshore zone was three times higher in 1970's compared to 1975 and 1979. In the midlake, phytoplankton biomass was significantly reduced between 1972 and 1983 (Munawar et al.,1987). Mean phytoplankton biomass was 43-58% less in 1982 than in 1972 (Gray, 1987). In 1970 approximately 3 g/m^3 was recorded which decreased ten-fold to 0.35 g/m^3 in 1983. Another measure of phytoplankton, particulate organic carbon, measured in the offshore upper waters in August/ September declined steadily from 1975 to 1981, with a 20% reduction occurring over this period (Dobson, 1984). Other workers have found no significant changes in particulate organic carbon, particulate organic nitrogen or chlorophyll a between 1974 and 1982 (Stevens and Neilson, 1987).

Various researchers have documented relationships between spring concentrations of TP and chlorophyll a and biomass. However, no significant relationship was found between spring or summer TP and algal biomass or chlorophyll a in Lake Ontario (Stevens and Neilson, 1988). The lack of relationship was attributed to the narrow range of concentrations of TP (within the same order of magnititude), whereas most models are based on a broader range of concentrations. Algal biomass is also affected by changes in phytoplankton community structure and predation effects.

Water clarity can be measured by Secchi disc or concentrations of particulates in a sample of water (mg/L). Water clarity measured by the Secchi disc usually follows an inverse relationship to the annual cycle of chlorophyll concentrations. Turbidity in water can be caused by suspended matter, plankton and microscopic organisms. Secchi disc measurements averaged over four cruises in 1981, were 4.2 m in the open lake beside the Niagara River and Rochester and 3.0 m near the Oswego River (Bertram, 1985). In the offshore waters of Lake Ontario, mean summer Secchi depths were 3.8 in 1965, 2.1 m in 1971, 3.4 m in 1977 and 2.2 m in 1982 (Dobson, 1984).

Eutrophication is considered an impairment of beneficial use in four of the AOCs on Lake Ontario: Hamilton Harbour, Metro Toronto, Bay of Quinte, and the Niagara River (Canadian shore). Eutrophication is not considered an impairment of beneficial use in Port Hope despite the fact that P concentrations exceed the OME's guideline in the turning basin and west slip (Port Hope RAP Team, 1989). In the Hamilton Harbour area, P, ammonia and nitrogen concentration exceed guidelines, and dissolved oxygen levels are lower than

guidelines. Algal blooms occur with corresponding problems in water clarity. As a further evidence of eutrophication, TP and TKN in sediment also exceed OME guidelines (Hamilton Harbour RAP Team, 1989).

In the Metro Toronto RAP, P concentrations also exceed guidelines, and *Cladophora* accumulations are a problem on the Western beaches. There are clarity problems from lakefilling. Nutrient levels in sediments also exceed guidelines (Metro Toronto RAP Team, 1989). In the Niagara River, the P concentration has decreased slowly and the N concentration has increased slowly since 1967. Turbidity problems occur in the Welland River. P concentrations exceed guidelines at the mouth of the Niagara River (Niagara River RAP Team, 1990). In the Bay of Quinte, P concentrations have decreased from the early 1970's, but still exceed guidelines. Algal growth has limited submerged macrophyte growth and may impair swimming, boating, aesthetics and ecosystem diversification (Bay of Quinte RAP Team, 1990). Water clarity has not been significantly reduced. For example, P loads from STP were 215 kg/day in the 1960's and are 32 kg/day in 1984. Blue-green algae still forms blooms in some sections of the Bay of Quinte.

5.8.6 SOURCES OF NUTRIENTS

Eutrophication is a result of excessive concentrations of nutrients, usually P and N. While estimates of their relative importance vary, the main sources of these nutrients are STPs, nonpoint sources such as fertilizer runoff from agricultural fields, industrial sources and atmospheric sources. A breakdown of the total P load to Lake Ontario is: 43% from Lake Erie; 30% from monitored tributaries; 18% from direct municipal sources; 7% from

unmonitored tributaries; 3% from atmosphere; and 0.5% from direct industrial sources (Stevens, 1988) (Table 5.8.1).

5.8.7 CONCLUSION

Average spring P concentrations in Lake Ontario have decreased significantly since the 1960s and have remained close to the target level of 10 μg/L since 1985. Nearshore zones generally have higher levels of P, and there are localized areas around STPs where concentrations can reach 25-30 μg/L. In contrast to phosphorus concentrations, nitrogen concentrations are increasing in Lake Ontario.

The phytoplankton response to these changes in nutrients has been decreasing biomass from 1970 to 1975. Since the mid-1970's algal biomass has remained fairly constant. Indicators of biomass such as particulate organic carbon, particulate organic nitrogen, and chlorophyll a have not significantly changed in Lake Ontario (Stevens and Neilson, 1982). Other workers have demonstrated a decrease in particulate organic carbon and biomass (Dobson, 1984). While nuisance blooms of *Cladophora* have been reduced, they are still a localized problem in many nearshore areas. Some of these areas such as Hamilton Harbour also exhibit depletion of dissolved oxygen concentrations in bottom waters and decreased water clarity. Therefore this impairment, eutrophication or undesirable algae, is occurring in Lake Ontario. The nutrients, phosphorus and nitrogen, which play a major role in controlling eutrophication should be considered causal pollutants.

Table 5.8.1. Summary of 1986 estimated atmospheric, industrial, municipal and tributary phosphorus loading data to the Great Lakes. All values are in tonnes/year.

	Superior	Michigan	Huron	Erie	Ontario	St. Lawrence River
Atmospheric[4] (standard error)	491 (104)	459 (80)	748 (119)	358 (56)	201 (29)	—
Direct IND Discharge (standard error)	81 (5)	60 (9)	23 (2)	38 (3)	29 (4)	14 (2)
Direct MUN Discharge (standard error)	78 (5)	367 (15)	134 (5)	1689 (67)	1384 (51)	146 (6)
Tributary: Monitored (standard error)	1345 (61)	3214 (139)	1845 (132)	6201 (224)	2376[1] (120)	119 (15)
Adjustment for Unmonitored Area (standard error)[2]	1064 (90)	881 (56	803 (202)	1753 (236)	652 (63)	90 (15)
Within Lake Totals	3059	4981	3553	10038	4642	370
From Connecting Channels	—	—	657	1080	4919	3087
OVERALL TOTALS (standard error)	3059 (151)	4981 (172)	4210 (269)	11118 (337)	9561 (291)	3456 (112)
Target Loads[3]	—	—	—	11000	7000	—

Totals may not sum due to rounding.

[1]Includes Buffalo River.
[2]Standard errors calculated from tributary loading estimates used in making adjustments.
[3]Annex 3, 1978 Great Lakes Water Quality Agreement.
[4]Atmospheric estimates were made from samples obtained from precipitation collectors, with any samples given a loading rate greater or equal to 0.4 mg/m^2d discarded as contaminated. Estimates are doubled to account for dry deposition.

Source: Great Lakes Water Quality Board (1989b).

5.9 RESTRICTIONS ON DRINKING WATER CONSUMPTION OR TASTE AND ODOUR PROBLEMS

5.9.1 DEFINITION AND DETERMINANTS OF DRINKING WATER IMPAIRMENT

The International Joint Commission (IJC) lists use impairment nine as "restrictions on drinking water consumption or taste and odour problems". Recently, the IJC (1991) published the following listing and delisting guidelines for impairment of this use.

Listing Guideline

When treated drinking water supplies are impacted to the extent that: 1) densities of disease-causing organisms or concentrations of hazardous or toxic chemicals or radioactive substances exceed human health standards, objectives or guidelines; 2) taste and odour problems are present; or 3) treatment needed to make raw water suitable for drinking is beyond the standard treatment used in comparable portions of the Great Lakes which are not degraded (i.e. settling, coagulation, disinfection).

Delisting Guideline

For treated drinking water supplies: 1) when densities of disease-causing organisms or concentrations of hazardous or toxic chemicals or radioactive substances do not exceed human health objectives, standards or guidelines; 2) when taste and odour problems are absent; and 3) when treatment needed to make raw water suitable for drinking does not exceed the standard treatment used in comparable portions of the Great Lakes which are not degraded (i.e. settling, coagulation, disinfection).

Drinking water intakes are located in nearshore areas which should not be directly influenced by point source or tributary discharges. Studies indicate that contaminants in raw water samples from Canadian drinking water intakes throughout the Great Lakes are generally at or below detection limits and are within existing water quality criteria and guidelines (Great Lakes Water Quality Board, GLWQB, 1989a). However, several years ago, Niagara-on-the-Lake, Ontario residents did change their source of drinking water from the Niagara

River to Lake Erie, due largely to Niagara River contamination by dioxins.

A 1977-78 survey estimated that the average consumption of tapwater-based beverages in Canada was 1.34 L/capita/day (Health and Welfare Canada, HWC, 1981). In the Toronto area, although drinking water meets existing standards, nearly one-quarter of all households use alternate sources such as bottled water or water filters. For Canada as a whole, a 1977-78 survey suggests approximately 8.5% of households have water softeners and approximately 3.7 % have water purifiers (HWC, 1981).

Impairment of water quality may result from bacterial contamination, nutrients, heavy metals, or organic chemicals which generally come from bypasses of sewage treatment plants, (STPs), as well as from combined sewer overflows and storm sewers. Storm sewers contain contaminants from surface drainage and anthropogenic sources; sanitary wastes originate from combined sewer overflow, especially during rainstorms.

5.9.2 POLICIES AND GUIDELINES

Drinking water standards, objectives, or guidelines are designed to ensure that any water intended for human consumption contains no disease-causing organisms or hazardous concentrations of toxic chemicals or radioactive substances. Aesthetic parameters such as temperature, taste, odour, and color also are controlled.

There are two major steps in the guideline setting procedure (Ontario Ministry of the Environment, OME, 1991).

1. **Hazard Assessment**

It involves qualitative and quantitative analysis to determine the potential effect a chemical could have in terms of health, safety, or environmental consequences.

2. **Risk Assessment/Management**

These take into account the Acceptable Daily Intake (ADI) or unit risk estimates calculated for a substance in order to develop a numerical limit for a substance in drinking water.

In the Province of Ontario, Drinking Water Objectives are comprised of three types of limits (OME, 1983).

1. **Maximum Acceptable Concentration (MAC)**

MACs apply to limits of substances above which there are known or suspected adverse health effects.

2. **Interim Maximum Acceptable Concentration (IMAC)**

IMACs describe limits for substances of current concern with known chronic effects in mammals and for which there are no established MACs.

3. **Maximum Desirable Concentration (MDC)**

MDCs refer to limits on substances which, when present at concentrations above the limits, are either aesthetically objectionable to an appreciable number of consumers or may interfere with good water quality control practices.

The Federal-Provincial Subcommittee on Drinking Water has designated guidelines for Canadian drinking water quality. The three types of limits are (HWC, 1989):

1. **Maximum Acceptable Concentration (MAC) (see above);**
2. **Interim Maximum Acceptable Concentration (IMAC) (see above);** and
3. **Aesthetic Objective (AO)** applies to certain substances or characteristics of drinking water that can effect its acceptance by consumers or interfere with practices for supplying good water.

Ontario has established forty-nine parameters for health and aesthetic objectives (currently under review). When an Ontario Drinking Water Objective is not available, guidelines/limits from other agencies are consulted. Ontario's Parameters Listing System (PALIS) consists of a handbook and a listing of parameters along with their corresponding guidelines which can be applied to drinking water. Each chemical is listed with the regulatory agency, type of water, guideline values, type and status and documentation from which the guidelines were extracted (OME, 1991). Ranges of PALIS limits are compared to Ontario Drinking Water Objectives (MACs, except MDCs for iron, zinc, and TDS) and Canadian Drinking Water Quality Guidelines (MACs, except AOs for iron, zinc, and TDS) for persistent toxic chemicals (designated in Annex I of the GLWQA) in Table 5.9.1.

In New York State, drinking water objectives are based on one type of numeric limit (NYCRR, 1990): the Maximum Contaminant Level (MCL) which refers to the maximum permissible levels of a contaminant. There are allowances for variances and exemptions from the MCLs in the guidelines.

Table 5.9.1. Drinking water parameters for persistent toxic chemicals[a] in Ontario waters. N/A indicates an objective is not available.

PARAMETERS LISTING CHEMICAL OBJECTIVE (PALIS)[e]	ONTARIO DRINKING WATER OBJECTIVE[b] (mg/L)	PALIS[c] (mg/L)	CANADIAN DRINKING WATER[d] (mg/L)
Aldrin/Dieldrin	0.0007	0.000003 - 0.0007	0.0007
Chlordane	0.007	0.000055 - 0.06	0.007
DDT	0.03	0.00001 - 1.4	0.003
Endrin	0.0002	0.000004 - 0.02	N/A
Heptachlor/Heptachlor Epoxide	0.003	0.000009 - 0.003	0.003
Lindane	0.004	0.0002 - 2.000	0.004
Methoxychlor	0.1	0.035 - 6.400	0.90
Mirex	N/A	N/A	N/A
Toxaphene	0.005	0.000005 - 1.400	N/A
Phthalic Acid Esters	N/A	N/A	N/A
Polychlorinated Biphenyls	0.003	0.000001 - 0.003	N/A
Arsenic	0.05	0.025 - 0.050	0.05
Cadmium	0.005	0.005 - 0.043	0.005
Chromium	0.05	0.050 - 1.400	0.05
Copper	1.0	0.01 - 1.3	1.0
Iron	0.3	0.050 - 0.500	0.3
Lead	0.05	0.010 - 0.100	0.01
Mercury	0.001	0.001 - 0.005	0.001
Nickel	N/A	0.0134 - 1.000	N/A
Selenium	0.01	0.001 - 0.045	0.01
Zinc	5.0	0.100 - 5.000	5
Fluoride	1.2	0.700 - 4.000	1.5
Total Dissolved Solids	500	100 - 500	500

[a]According to Annex 1 of GLWQA (1989); [b]OME (1983); [c]OME (1991); [d]HWC (1989); [e]OME (1990).

Guideline values in Ontario describe a quality of water that is acceptable for lifelong consumption. Therefore, short-term deviations above the guideline values do not necessarily mean that water is unsuitable for consumption. U.S. EPA health advisories, on the other hand, are designed for the case of an emergency spill or acute exposure.

Not all chemicals identified in drinking water throughout the world can be targeted for guidelines. Limits established by agencies have been derived from the best information currently available; the development of objectives is an on-going process requiring continual revision.

5.9.3 HEALTH CONSIDERATIONS

Ontario Drinking Water Objectives list four categories of water quality characteristics related to health considerations (OME, 1983).

1. **Chemical.** Toxic chemicals can affect human health. Thus heavy metals and ions, some commonly occurring organic compounds and many less common organic and organometallic substances are potentially hazardous.
2. **Physical.** Although this includes color, odour, taste, temperature and turbidity, only turbidity has a limit set on the basis of health considerations.
3. **Microbiological.** Typhoid fever, cholera, enterovirus disease, bacillary and amoebic dysenteries, and many varieties of gastrointestinal disease can be transmitted by water.
4. **Radioactivity.** Radiation doses to organs and tissues can result from intakes of radionuclides.

It is clear that the primary purpose of Drinking Water Objectives is the protection of human health.

In addition to Ontario's Drinking Water Surveillance Program, two studies with respect to health considerations and water quality in the Lake Ontario basin were identified. In the most recent study (Palmateer et al., 1990), eighteen water samples from raw and treated drinking water from three Ontario cities were analyzed; for two of these cities the drinking water came from one of the Great Lakes. In each instance, total coliform, fecal coliform and fecal streptococcus counts were not detected in 100 mL water samples. Coliphage and bacteriophage were present before and after treatment, which suggests that viruses can also survive the normal treatment and disinfection processes.

The second and more dated study investigated Lake Ontario water quality in the area of Port Hope, Ontario (Durham and Joshi, 1980). Results indicated that ^{226}Ra, As, and NO_3 have leached from the Port Granby waste management site of Eldorado Nuclear Limited reaching Lake Ontario. Elevated levels of all three substances were measured in samples of inshore water close to the site, but the radioactivity objective of the Great Lakes Water Quality Agreement was exceeded only by two samples (Durham and Joshi, 1980). No samples of treated water were taken.

5.9.4 AESTHETIC CONSIDERATIONS

The primary goal of aesthetic considerations and drinking water is to produce a treated water that pleases consumers, in the hope of discouraging the use of alternative water sources (which may be aesthetically pleasing, but may not have adequate standards with respect to health-related parameters).

Ontario Drinking Water Objectives list three categories of water quality characteristics related to aesthetic considerations (OME, 1983).

1. **Chemical.** Limits are given for eight inorganic chemical species and high levels of organic matter tend to be associated with color, taste, and odour problems.

2. **Physical.** Although this includes color, odour, taste and temperatures, numeric limits are only listed for color and temperature.

3. **Biological.** Many of the problems caused by nuisance organisms are covered by the limits on the physical characteristics of water.

In Ontario, personnel of water-treatment plants evaluate taste and odour water quality parameters. There are no standardized methods of detection.

In the United States, organoleptic quality is routinely monitored with the threshold odour number (TON) method. The TON is defined as the largest dilution factor of a sample that still yields a definitely perceptible odour. Dilution is done with odour-free water (blank). This method does not include a qualitative description of the perceived taste and odour.

Detection of odour requires that a substance be volatile and able to reach the olfactory nerve receptors in the uppermost recesses of the nasal cavities. Part of the so-called taste of water is, in fact, a smell sensation, which only develops after the water is swallowed. Not only do chemicals vary immensely in their detectability by the sense of smell and in the quality of odour they possess, but people differ a great deal in their sensitivity to odours. The study of odour thresholds has revealed that many people have a selective "smell-blindness" to just one class of odours and may be up to 1,000 times less sensitive than "normal" individuals. Conversely, some disorders, especially Addison's disease and certain pituitary tumors, may actually be accompanied by a sensitivity to odours that is more than 1,000-fold greater than normal.

There is no standardized categorization of sources of organoleptic problems. Bartels et al. (1986) categorized sources of organoleptic problems into four groups.

1. Distaste and odour are often caused by algae and microbial metabolites.

2. Industrial chemicals may be responsible such as chlorobenzenes or alkanes.

3. During water treatment, especially chlorination, trace organics can be converted into compounds that have an objectionable taste and odour.

4. New compounds can be added from the lining of water distribution pipes or storage facilities.

Evidence supports the general agreement that the sense of smell operates on the basis of a limited number of discrete "primary odour" sensations, which can be combined in different proportions to give a tremendous range of distinguishable odours. Olfactometric properties of eight of the primary odourants are given in Table 5.9.2. The primary odourants are those for which the sense of smell is particularly acute. Compounds, whether man-made or natural, that are closely related to the primary odourants and occur in drinking water supplies are likely to be noticed and cause complaints.

Brownlee et al. (1984) analyzed the water supply of Burlington, Ontario, in the summer of 1983. They found geosmin (trans, trans-1, 10-dimethyl-9-decalol) in concentrations of 0.01-0.07 μg/L. The odour detection threshold for geosmin is in the range 0.01-0.2 μg/L. At the time, there was a massive die off of *Cladophora*. It is hypothesized that geosmin was produced in the water-treatment plant by actinomycetes (bacteria), possibly from organic matter released during the *Cladophora* degradation; sophisticated microbiological assays are needed to detect this and were not available.

Table 5.9.2. Olfactometric properties of eight of the primary odourants.

		Normal Threshold	
Primary Odourant	Primary Odour	In air (μL/L)	In water (mg/L)
Isovaleric Acid	Sweaty	0.0010	0.12
1-Pyrroline	Spermous	0.0018	0.020
Trimethylamine	Fishy	0.0010	0.00047
Isobutyraldehyde	Malty	0.0050	0.0018
5α-Androst-16-en-3-one	Urinous	0.00019	0.00018
ω-Pentadecalactone	Musky	0.018	0.0018
2-Carvone	Minty	0.0056	0.041
1,8-Cineole	Camphorous	0.011	0.020

Source: Amoore (1982).

5.9.4 OTHER CONSIDERATIONS

In a recent report (Great Lakes United, GLU, 1988), a number of concerns were expressed that may impact on drinking water quality. (1) While drinking water-treatment plants effectively eliminate harmful bacteria, most were not designed to remove persistent toxic chemicals. (2) Even though concentrations of some persistent toxic chemicals in lake water have diminished in the last decade, the rate of decrease may have slowed. (3) The impact of aging dumpsites may be expected to increase in spite of a reduction in direct discharges.

Trihalomethanes (THM) are formed when chlorine used to disinfect raw water as it enters the sewage treatment plant combines with trace levels of organics found in the raw water. Turbidity is related to THM formation. Although the health risk of the THM levels needs further research, alternative methods of water treatment, including ozonation and activated carbon filtration, are currently being assessed.

Questions have been raised with respect to the adequacy of drinking water quality guidelines. These include the criteria used to establish allowable levels, the fact that guidelines do not cover all the chemicals found in drinking water and the assessment of health hazards themselves. There is insufficient information available on the potential health effects of many chemicals.

Few municipalities have established emergency procedures or alternative sources of drinking water should the worst case scenario such as a spill happen in Lake Ontario (GLU, 1988).

Analytical methods used for drinking water analysis should be reviewed. In 1979, in the United States, a set of "500 series" laboratory-based analytical methods focusing on drinking water was developed. The methods reflect various levels of analytical technology because of the different times at which the methods were developed. There are also differences in the detection limits of these methods. Summary recommendations from a recent follow-up study of 24 methods from the 500 series included (Hites and Budde, 1991):

(i) some existing 500 series methods should be revised and others should be dropped;

(ii) format of official EPA methods should be standardized throughout the Agency, and the writing of revised methods should be modularized;

(iii) a usage survey of EPA's analytical methods should be conducted; and

(iv) revised or new methods should have expiration dates.

5.9.5 SURVEILLANCE PROGRAMS

In Ontario, the Drinking Water Surveillance Program (DWSP) provides immediate, reliable and current information on drinking water quality. Currently, raw, treated, and distributed water at 44 water supply systems are being monitored (monthly). Analyses for up to 180 parameters are carried out on each water sample, including microbiological, organic and inorganic substances as well as process parameters (OME, 1989c).

The most recent DWSP Overview Annual Report, published in 1989, presents a summary of the DWSP for 1987. Most supplies had observed concentrations that were higher than the aesthetic Ontario Drinking Water Objectives for nitrogen and temperature. At most treatment plants where alum is used as a primary coagulant, the recommended guideline level for residual alum in water entering the distribution system was exceeded. Recommended levels of fluoride were not maintained in all of the supplies. Positive organic occurrences for chloroaromatic compounds, pesticides, and volatiles were found in some plants using Lake Ontario as a source. The levels of these organic compounds found, while quantifiable, were relatively low, and all levels were well below any available health-related guidelines. Although benzene concentrations were very low, generally below detection limits, its concentration is expected to increase with time due to its use as an additive to gasoline (replacing lead). No polychlorinated biphenyls (PCBs) were found in any samples of treated or distributed water. No health-related Ontario Drinking Water Objectives for metals were exceeded. Some DWSP results for plants using Lake Ontario as a source are presented in Table 5.9.3. The water quality produced by all of the water supply systems on the DWSP for 1987 was generally good. The monthly sampling frequency of the DWSP is not considered to generate enough data for a full evaluation of the waters surveyed.

Non-standard treatments of raw water were not required to achieve acceptable water quality.

Table 5.9.3. Overview of drinking water surveillance program (DWSP) results in Ontario for 1987[a].

Location	Type of Treatment	Population Served	Level of Occurrence[b]
Grimsby	Conventional	15,000	· organic nitrogen (1) · temperature (1) · endosulfan sulphate (2)
Hamilton	Conventional	412,000	· fluoride (1) · organic nitrogen (1) · temperature (1) · benzo (A) pyrene (2) · bladex (2) · hexachloroethane (2) · prometone (2) · simazine (2)
Burlington	Direct Filtration	120,000	· fluoride (1) · organic nitrogen (1) · temperature (1)
South Peel (Lakeview)	Conventional	450,000	· fluoride (1) · organic nitrogen (1) · benzene (2) · m-xylene (2)
South Peel (Lorne Park)	Conventional	450,000	· fluoride (1) · organic nitrogen (1)
Metro Toronto (R.L. Clark)	Conventional	2,333,000[d]	· fluoride (1) · organic nitrogen (1) · temperature (1) · coliform (2) · 1,1,1-trichloroethane (2)
Metro Toronto (R.C. Harris)	Conventional	2,333,000[d]	· fluoride (1) · iron (1) · organic nitrogen (1) · temperature (1)
Metro Toronto (Easterly)	Direct Filtration	2,333,000[d]	· coliform (2)

Table 5.9.3 Continued.

Location	Type of Treatment Served	Population	Level of Occurrence[c]
Kingston	Conventional	81,000	· organic nitrogen (1) · temperature (1) · hexachloroethane (2)
Oshawa	Conventional	244,000	· fluoride (1) · organic nitrogen (1) · temperature (1) · turbidity (1)

[a]Source: OME (1989c).
[b]Values that exceeded Ontario Drinking Water Objectives are denoted by (1). Positive occurrences are denoted by (2).
[c]Because the number of samples tested varied for each substance and in instances was small, frequency occurrene is not reported. The reader is referred to the original document for this information.
[d]R.L. Clark, R.C. Harris, and Easterly regions of Metro Toronto collectively serve a population of 2,333,000.

In New York State, there was no information available with respect to a state-wide drinking water surveillance program. Average results of surveyed parameters from Monroe County Water Authority were available but no recorded values exceeded water quality limits.

5.9.6 CONCLUSIONS

Based on the Ontario Drinking Water Surveillance data reported in Table 5.9.3, it is clear that organic nitrogen consistently exceeds the Ontario Drinking Water Objectives. Consequently, there appears to be grounds for considering organic nitrogen as a causal pollutant. As summarized in Section 5.8, it also contributes to eutrophication of lake water.

5.10 BEACH CLOSINGS

5.10.1 DEFINITION AND DETERMINANTS OF BEACH USE

The tenth use impairment, "beach closings", implies restrictions imposed not only on recreational swimming, but on diverse human activities. Recently, the International Joint Commission (IJC, 1991) published the following guidelines for listing and delisting this impairment of beneficial use.

Listing Guideline

When waters, which are commonly used for total-body contact or partial-body contact recreation, exceed standards, objectives, or guidelines for such use.

Delisting Guideline

When waters, which are commonly used for total-body contact, or partial-body contact recreation, do not exceed standards, objectives, or guidelines for such use.

Beach use itself was categorized by Usher et al. (1987) as a combination of primary water-oriented activities and primary non-water oriented activities. The former consisted of contact activities (swimming, skin/scuba diving, waterskiing, and windsurfing), and non-contact activities (fishing, boating, and sailing). Primary non-water-oriented activities included those classified as non-contact, beach oriented (sunbathing), and non-contact, outdoor oriented (recreational walking, natural and cultural heritage appreciation, picnicking, and casual outdoor sports).

A discrete division is made between "guidelines determining beach use" and "factors determining the use of beaches". The former provides the basis for "official" beach closures and/or postings (i.e., swimming restrictions), whereas the latter tends to indicate and perhaps

dictate use level of the beach area itself. Usher et al. (1987) presented a list of five categories of factors determining the use of beaches.

1. **Aquatic (biological)**

   - Including the parameters of color, taste, odour, turbidity/clarity, pH, indicators of bacterial contamination, pathogens, toxins, filamentous algae, blue-green algae, odour, weeds, oil/grease/scum/foam, floating/beached objects, insects/parasites, nuisance birds.

2. **Physiographic**

   - Including lake, beach length, wet beach width/slope, dry beach width, beach composition, back-shore conditions, exposure, current/undertow.

3. **Climate**

   - Including water temperature, air temperature, sunlight, wind.

4. **Development and Management**

   - Including development, aesthetics/intrusions, ancillary facilities/opportunities, parking, incompatible recreational activities, admission charges, management character and intensity.

5. **Social and Economic**

   - Including long-term regional demand characteristics, travel times/tributary population, travel ease, travel modes, availability of suitable sites, day of week/time of day.

The parameters within these categories may then, either alone or in some combination, play a role in the definitive use impairment (even though they may not be incorporated into jurisdictional guidelines).

5.10.2 WATER QUALITY INDICATORS USED FOR BEACH CLOSINGS

There are six water quality indicators considered for decisions on beach closings by various jurisdictions.

1. Bacterial Indicators

The human health risk (based on bacteriological densities) in polluted beach waters depends upon the presence of pathogens including bacteria, fungi, protozoa, and viruses carried by the water or resuspended from the bottom sediment by the bathers and/or wave action. Contact with these pathogens may cause infections of the skin, gastrointestinal tract, or upper respiratory system.

Most commonly, fecal coliform (found in fecal wastes from warm-blooded animals and birds) and total coliform density measures are used as indicators of risk for human health. In Table 5.10.1, the range of bacterial indicators for pollution used by various jurisdictions are summarized. *Escherichia coli* alone is the most reliable indicator of fecal pollution and may better serve to provide public health authorities with a measure of potential risk of gastointestinal disease. The percentage of *E. coli* normally found in the fecal coliform test routinely ranges from 63 to 100 percent in recreational waters impacted by fecal wastes (Toth, 1990).

Table 5.10.1. Bacterial indices of water pollution.

Bacterial Indicator	Consequences of Exposure	Frequency of Use for Monitoring
Escherichia coli	Diarrhoea, urinary infection	frequent
Klebsiella spp.	Urinary infection, sepsis	occasional
Pseudomonas aeruginosa	Swimmer's ear	seldom
Enterobacter spp.	Urinary infection (low)	occasional
Citrobacter spp.	Urinary tract infection	occasional
Streptococci	Ear, nose, throat infection	occasional

Source: Sleigh and Timburg (1986).

While nearly all causes for beach postings have been associated with high fecal coliform counts, it is essential to realize that identification of impairments to this beneficial use incorporates both objective and subjective criteria.

2. Sanitary Survey

In conjunction with bacterial levels at specific sites, many jurisdictions use sanitary surveys as indicators for impairments to water quality. Survey parameters are listed below (Chong, 1984):

(i) the risk of inadequately treated sewage, fecal matter or other hazardous substances entering the water;

(ii) all outfalls or any drainage that may contain sewage in the area, particularly upstream;

(iii) on-site inspections by competent observers or inspectors who are knowledgeable of the area;

(iv) establishment of a reporting system when in case of an environmental emergency the appropriate authorities can be contacted promptly;

(v) rainfall;

(vi) peak use; and

(vii) sources from other jurisdictions.

3. Physical Quality of Water

Aesthetic indicators for qualities/impairments of the beach area include presence of dead fish, unpleasant odours, algal blooms, and physical debris.

4. Epidemiology Data and Disease Outbreak Reports

5. Information Concerning other Potential Indicator Organisms

6. Past or Present Bacteriological Data

5.10.3 POLICIES AND GUIDELINES

1. Water Quality Indicator Sampled

The Ontario Ministry of the Environment (OME) and the Ontario Ministry of Health (OMH) are currently involved in re-evaluating recreational water quality objectives.

In the jurisdictions that sample beach water, the most frequently monitored bacterial density is fecal coliform which is expressed as fecal coliform count per 100 mL (fc/100 mL). Several jurisdictions routinely measure densities of fecal *Streptococci* and *Pseudomonas aeruginosa*. Often a combination of fecal coliform and total coliform densities are presented. Total coliform density on its own is not used as a water quality indicator.

The use of the fecal coliform density as a recreational objective is controversial because the presence of high *Klebsiella* spp. densities are known to influence and upset the usual fecal coliform test result interpretation. The Federal-Provincial Working Group on Recreational Water Quality recently recommended that the federal government endorse an *E. coli* objective (E. Leggatt, Ontario Ministry of the Environment personal communication).

In the United States, the Environmental Protection Agency (EPA) proposed the use of *E. coli* and *Enterococci* spp. densities as water quality indicators. New York State concluded that although useful, these tests for pathogens were inadequate to establish a linear correlation to gastrointestinal illness (Svenson, 1990).

2. Criteria for Beach Closure

Both New York State and Ontario have policies for beach closing. The Ontario Ministry of Health policy (OMH, 1975) states:

> *The quality of bathing beach water is considered impaired when total coliform geometric mean exceeds 1000 per 100 mL and/or fecal coliform geometric mean exceeds 100 per 100 mL. Total coliform counts should not be considered alone.*

The Federal-Provincial Working Group on Recreational Water Quality (FPWGRWQ, 1983) states:

> *The maximum limit set is the geometric means of not less than 5 samples taken over a 30 day period. These should be less than 200 fc/100 mL. No maximum acceptable limit has been set for total coliforms.*

In New York State, a criteria of 200 fc/100 mL is used. A local health officer may also decide to post a beach for aesthetic or safety reasons (Great Lakes Water Quality Board GLWQB, 1989b).

Recent EPA proposed legislation (Svenson, 1990) requires multi-criteria monitoring and indicates the individual States are:

(i) to be consulted by EPA in its review and development of proposed standards; and

(ii) will be given the option of adopting their own standard as long as it meets a uniform set of criteria or objectives.

3. Sampling Frequency

In Ontario, there are two guidelines for sampling frequency.

(i) A series of at least 10 samples per month per sampling location with increased sampling frequency for recreational purposes is recommended (OME, 1978; OMH, 1975); and

(ii) A minimum of 5 samples per month from each area; increased frequency at beaches with higher bather densities is suggested (FPWGRWQ, 1983).

The OMH suggests using a geometric mean of fecal coliform densities based on the results of 10 fecal coliform samples as the criteria to post beaches. Most units sample 5 days per week from 5 fixed, equidistant sampling stations per beach.

No information was available with respect to New York State beach water sampling frequency.

4. Sampling Methodology

For each sampling occasion, the OME recommends samples be taken at water depths of 1-1.5 m 15-30 cm below surface at the time of peak swimming activity with a minimal sampling frequency of 12 samples per area. The actual method of sampling is not stipulated.

A 1984 survey (Chong, 1984) found that more than one method of sample collection is used by the majority of health units in Ontario. Some methods may involve a higher risk of sample contamination than others. Even with similar sampling methods, both the precision and accuracy varied. Sources of variation included the wide range of lengths of reaching poles used. In most cases, bacterial tests are performed by OMH Laboratories. In the case of Halton, Peel, Etobicoke and Toronto, however, bacterial testing is done by OME Laboratories.

In New York State, as many as three levels of government are involved in monitoring. These include municipal, county and State Health and State Park Authorities. There are serious deficiencies when it comes to data reporting, data quality, and data uniformity (GLWQB, 1989). No information addressing New York State sampling methods was available. In the eastern Lake Ontario/Cape Vincent area, there has not been any recent sampling due to a lab closure at Alexandria Bay (not expected to reopen).

5.10.4 BACTERIOLOGICAL TRENDS

Bathing beach monitoring for bacteria is required by the OMH and the NYSDH. The actual data collections and analyses are carried out by Regional/County Health Units. In both situations, no centralized collection of information exists, making impact assessment difficult.

A study of the bacteriological quality of Ontario beaches indicated that the general status of Lake Ontario shoreline beaches between 1972 and 1982 was classified as "stable to improving" (Table 5.10.2; Chong, 1984).

Table 5.10.2. Bacteriological quality of Ontario beaches, 1972-1982.

Health Agency	Bacteriological Quality
Haliburton, Kawartha, and Pine Ridge	improving
Hastings and Prince Edward	improving
Durham	stable
Scarborough	improving
Toronto	stable
Etobicoke	degrading
Peel	improving
Halton	improving
Hamilton-Wentworth	stable
Niagara	stable

Source: Chong (1984).

Bacteriological data collected at 55 locations along the Toronto waterfront between 1967 and 1980 have been analyzed for trends (Envirosearch, 1981). A direct relationship was found between precipitation and bacterial quality at sewer outfalls along the waterfront. Although the available data showed considerable variability, the degree and duration of the contamination following rain events was dependent upon prevailing winds and generated currents.

Recent beach closure data were obtained for this study from all Lake Ontario shoreline Ontario Health Units and analyzed for trends in bacterial contamination. Summary data from the Toronto Public Health Unit (the center of previous studies) is illustrated in Table 5.10.3. A summary of the number of days posted and expressed as a percentage of total sampling for the time period 1985 to 1990 is provided. Similar information was requested from the

Table 5.10.3. Six-year summary of Toronto Waterfront beach postings (1985-1990)[a]. Data shown are the number of days and the percent of the total days sampled when the beach was posted.

Beach	1985 # DAYS (POSTED %)	1986 # DAYS (POSTED %)	1987 # DAYS (POSTED %)	1988 # DAYS (POSTED %)	1989 # DAYS (POSTED %)	1990 # DAYS (POSTED %)
Western:						
Windermere	N/A	N/A	67 (68%)	59 (61%)	70 (81%)	78 (89%)
Ellis Avenue	57 (55%)	57 (52%)	67 (68%)	59 (61%)	70 (81%)	78 (89%)
Sunnyside	48 (56%)	47 (43%)	67 (68%)	59 (61%)	70 (81%)	78 (89%)
Boulevard Club	27 (26%)	33 (30%)	67 (68%)	59 (61%)	70 (81%)	72 (83%)
Eastern:						
Balmy Beach	25 (23%)	42 (40%)	56 (58%)	44 (45%)	0 (0%)[b]	7 (8%)
Beaches Park	22 (21%)	21 (20%)	42 (43%)	44 (45%)	0 (0%)[b]	7 (8%)
Kew Beach	22 (21%)	25 (24%)	42 (43%)	44 (45%)	0 (0%)[b]	7 (8%)
Woodbine Beach	20 (19%)	23 (22%)	33 (34%)	33 (34%)	0 (0%)[b]	7 (8%)
Islands:						
Centre Island	32 (31%)	28 (25%)	55 (57%)	40 (41%)	47 (55%)	38 (44%)
Olympic Island	33 (31%)	38 (35%)	55 (57%)	40 (41%)	36 (42%)	42 (48%)
Snake Island	0 (0%)	18 (17%)	55 (57%)	40 (41%)	28 (33%)	34 (39[illegible]
Cherokee	43 (41%)	41 (38%)	47 (48%)	40 (41%)	21 (22%)	36 (41%)
Ward's Island	29 (28%)	33 (30%)	25 (26%)	20 (21%)	14 (16%)	34 (39%)
Hanlan's Point	1 (1%)	23 (21%)	15 (15%)	33 (34%)	29 (34%)	36 (41%)
Cherry Beach	0 (0%)	7 (6%)	22 (25%)	26 (27%)	0 (0%)	0 (0%)
Total # of Sampling Days:						
	105 days (Western & Island = 104 days)	105 days (Western & Island = 109 days)	97 days (Western = 99 days)	97 days	86 days	87 days

[a] Source: Toronto Department of Public Health (TDPH, 1991).
[b] Not officially posted due to 48-h rainfall rule (i.e. beaches considered polluted for 48 h after rainfall).

remaining Lake Ontario shoreline health units. When this type of a summary was not available (which was generally the case), geometric means of fecal coliform densities were forwarded.

Although the majority of Ontario Health Units responded, the type of information received was so diverse (Table 5.10.4) that a trend assessment was difficult. A general conclusion was that beach quality on the Lake Ontario shoreline appeared to be in a deteriorating state in contrast to what Chong (1984) found for the 1972 to 1982 period (see above). Not only did the type of data vary, there was also dissimilarity in the criteria used for beach closure. While the majority of health units used the Provincial guidelines of 100 fc/100 mL, there was at least one case where the judgement was based on a limit of 200 fc/100 mL. The latter is the level suggested by the Federal-Provincial Working Group on Recreational Water Quality.

The situation along the Lake Ontario shoreline in New York State presents a similar scenario. Individual County Health Departments do not report beach data to the State Health Department headquarters in Albany. Information presented by the International Joint Commission (Table 5.10.5; GLWQB, 1989b) suggests a relatively stable situation from 1980 to 1985 based upon beach closures.

5.10.5 SOURCES OF BACTERIAL CONTAMINATION

According to policies and guidelines currently in effect, the primary water quality indicator used in the determination of beach closings is the presence of bacteria, specifically fecal coliform. Factors which control this indicator are not easily defined, but are suspected

Table 5.10.4. Summary of information – Canadian Health Units along Lake Ontario with public beaches.

Health Unit	# of Beaches (year figure confirmed)	Fecal Coliform Density Guidelines Used	Type of Data Received	Comments
Haliburton, Kawartha, Pine Ridge	5 on L. Ont. Shoreline (1991)	not given	verbal statement (1991)	• 1990: first time in many years that all L. Ont. shoreline beaches were posted
Hastings and Prince Edward	16[a] (1990)	not given	1990 beach area annual geometric means; date posted/ unposted; high count; avg. mean; # of samplings	• Hastings (1990): majority of beaches closed for entire season. • P.E. (1990): majority of beaches closed for approx. 1 week; generally in later part of summer.
Durham	17 on L. Ont. Shoreline (1984-1990)	not given	1984-1990 annual avg. as geometric means	• acceptable annual avg. geometric means until 1990. • 1990: all avg. geometric means exceed 100 fc/100 mL. • 1990: 9 of 17 beaches posted.
Scarborough	2 (1991)	not given	verbal statement (1991)	• 1990: both beaches posted
Toronto	15 (1985- 1990)	100 fc/100 mL	1985-1990 summary of postings: # of days posted; (%)	• see Table 5.10.3
Etobicoke	3 (1985-1990)	100 fc/100 mL	1985-1990 summary of postings: # of days posted; (%)	• no visible trends
Peel	8 (1990)	not given	verbal statement (1991)	• 1990: all beaches closed July 31 and remained closed for entire season
Halton	10 (1990)	200 fc/100 mL	1989-1990 monthly avg. as geometric means	• avg. monthly means consistently exceed 100 mL. Aug. 90 considerably higher means than Aug. 89
Hamilton-Wentworth	11 (1983)	100 fc/100 mL	1985-1990 summary of postings; # of days posted; (%)	• no visible trends
Niagara (specifically St. Catharines)	4 (1991)	100 fc/100 mL	verbal statement (1991)	• All area beaches closed entire season for past 15 years. OME stated that some improvement has occurred.

[a]All may not be Lake Ontario Shoreline.

Table 5.10.5. Lake Ontario beach closures in New York State.

Year	Beaches Surveyed	Beaches Restricted	Beaches Closed
1980	22	5	2
1981	23	4	3
1982	24	3	2
1983	24	3	2
1984	26	3	2
1985	20	3	2

Source: GLWQB (1989b).

to be rainfall and wind/wave related.

Lake Ontario acts as the terminal receiver for many effluent discharges as well as direct discharges in the basin area. After a review of several documents, what follows represents a summary of the primary sources of bacterial contamination of Lake Ontario shoreline beaches:

1. Stormwater Runoff

In many areas, stormwater runoff is discharged directly into Lake Ontario. There is a need to address pollutant sources in areas in which stormwater discharges in the vicinity of recreational beaches. A report from St. Catharines, Ontario (Canviro, 1989a) proposes three control alternatives for stormwater runoff:

(i) Source Quality Controls
- includes enhanced street sweeping, catch basin cleaning, and stronger anti-litter bylaw enforcement;

(ii) Dispersion/Diversion

- involves the extension of stormwater outfalls (fitted with multi-port diffusers) into Lake Ontario as well as the installation of shoreline interceptors, i.e., diversion sewers; and

(iii) Storage/Treatment

- stemming from the goal of natural die-off of fecal organisms.

2. Combined Sewer Overflow (CSO)

Many beach-area water-quality problems are due to the release of untreated sewage effluent via CSOs to surface waters during storms and significant rainfall events. Many studies have been done, some initiating remedial programs designed to control this problem. The St. Catharines, Ontario, Area Pollution Control Plan (SCAPCP) proposes infrastructure improvement programs (OME, 1990b) which include flow reduction (via on-lot drainage separation, road drainage separation, and infrastructure repair) as well as flow control (via storage and pipe/pump repairs) (Toth, 1990).

Monroe County in New York State has a Combined Sewer Overflow Abatement Program (CSOAP) currently in place. The program, which is under the umbrella of the 1969 Pure Waters Master Plan, consists of construction of huge underground storage/conveyance tunnels, additional treatment facilities, and a program of best management practices. This program is funded by all three levels of government.

3. Dry Weather Seepage

Programs to remedy this would be similar to those established to control stormwater runoff with particular emphasis on source quality controls.

4. Sewage Treatment Plants (STPs)/Water Pollution Control Plants (WPCPs)

In the Province of Ontario, there are some 412 sewage treatment plants taking in not only sewage but also waste water from approximately twelve thousand industries. Of the 412 facilities, the larger ones are primarily located in the Lake Ontario drainage basin, accounting for 58.4 percent of the total flow from Ontario STPs and WPCPs (Canviro, 1989b).

The initiative of Ontario's Municipal-Industrial Strategy for Abatement (MISA) Program is to establish an abatement schedule for all of the major specific toxic pollutants. The goal is virtual elimination of both municipal and industrial toxic discharges into waterways (OME, 1986). A "37 STP Survey" (Canviro, 1989b) has been completed as part of the municipal sector of the MISA Program. The survey consisted of a comprehensive field monitoring program which will serve to form the basis for the establishment of discharge standards.

In the United States, the Clean Water Act declares it unlawful for anyone to discharge pollutants from a point source to waters of the United States without a permit, (NRTC, 1984). In New York State, this permit program is named State Pollutant Discharge Elimination System (SPDES).

5. Industrial Discharge

Although beaches have not been closed for non-bacterial reasons (e.g., contaminants, oil, grease, etc.), industrial effluent often provides a rich habitat conducive to bacterial growth. High nutrient levels and neutral pH enhance bacterial survival (Beak, 1989). These characteristics are often present in industrial discharges and thus discharge reduction may well help to alleviate bacterial contamination. As effluent is diluted by receiving water, conditions

become less favourable for bacterial survival.

In Ontario, the MISA Program is a response to the problem of industrial discharge. MISA encompasses nine industrial sectors (OME, 1986):

(i) electric power generation;
(ii) industrial minerals;
(iii) inorganic chemicals;
(iv) iron and steel;
(v) metal mining and refining;
(vi) organic chemicals;
(vii) petroleum refining; and
(viii) pulp and paper.

Recently a ninth sector, metal casting, was added.

These eight sectors comprise 200 of 300 direct dischargers. To date, cost estimates and implications of effluent monitoring have been obtained for the industrial minerals sector (OME, 1989a), the petroleum refining sector (OME, 1988a) and the inorganic chemicals sector (OME, 1989b). There are two approaches in the setting of the effluent limits:

(i) the best available technology economically achievable (BATEA); and
(ii) water quality impact.

It is MISA's mandate that the discharger will be required to comply with the more stringent of the two limits (OME, 1986). The MISA BATEA effluent limit approach is based on the work of the EPA, which has carried out extensive technical development of effluent limits based on various levels of technology including:

(i) the costs of applying the control technology;
(ii) the age of process equipment and facilities;
(iii) the process employed;
(iv) the process changes;
(v) the engineering aspects of applying various types of control technologies; and
(vi) non-water quality environmental considerations.

Note that the approach does not require a balancing of costs against effluent reduction

benefits (OME, 1986). In addition to a number of treatment-before-discharge technologies, BATEA may include process changes in the plant, substitution of chemicals, in-plant treatment, recycling and reduced water usage.

In Monroe County, New York State, the Department of Pure Waters conducts a rigorous industrial discharge permit program under the authority of the County's Sewer Use Law for industries discharging their effluent to the County's sewage system (Peet, 1990).

Statewide SPDES permits control discharges from industrial and municipal point services (NRTC, 1984). The permits address toxic chemicals which are known or are suspected to be in the discharge, based on information on chemicals used and plant processes. The permit limits are established on three considerations (NRTC, 1984):

(i) natural technology standards where available (Best Available Technology Economically Feasible or Best Engineering Judgement where natural standards do not exist);

(ii) the effect of the discharge on the best use of the receiving waters as reflected on ambient water quality criteria or standards; and

(iii) a reasonably achievable detection limit for the chemical in the discharge.

5.10.6 CONCLUSIONS

All sources of nutrient contamination discussed provide habitats conducive to bacterial propagation. When elevated bacterial levels occur, beach use is impaired. It is concluded that bacteria is the causal agent for the use impairment beach closing.

5.11 DEGRADATION OF AESTHETICS

5.11.1DEFINITION AND DETERMINANTS OF AESTHETICS

Perhaps one of the most difficult impairments to qualify and quantify is the eleventh, "degradation of aesthetics". Listing and delisting guidelines for this impairment are shown below (International Joint Commission, IJC, 1991).

Listing Guideline

When any substance in water produces a persistent objectionable deposit, unnatural colour or turbidity, or unnatural odour (e.g., oil slick, surface scum).

Delisting Guideline

When the waters are devoid of any substance which produces a persistent objectionable deposit, unnatural colour or turbidity, or unnatural odour (e.g., oil slick, surface scum).

Our review of this beneficial use impairment initially focussed on recognized aesthetic problems including those related to odour. It is clear from the material presented that very little concrete information is available on this topic. To complete our survey, we include some findings of the Royal Commission on the Future of the Toronto Waterfront (RCFTW; the Crombie Commission). In the Commission's analysis and recommendations, aesthetics of the waterfront are linked to development and use. Some general discussion of how changes in aesthetics are observed and perceived is also included.

5.11.2 AESTHETIC QUALITY OF BEACH AREAS

The results of a 1983 survey which identifies aesthetic problems at selected Lake Ontario beach areas (Chong, 1984) are summarized in Table 5.11.1 The most common

Table 5.11.1. Aesthetic problems at some Lake Ontario beaches[a].

Health Unit	Type of Problem	Severity
Haliburton, Kawartha and Pine Ridge	· nil	
Hastings and Prince Edward	· nil	
Durham	· dead fish · *Cladophora* sp. · debris	· moderate
Scarborough	· heavy pollution from Rouge River · algal bloom	· slight
Toronto	· dead fish · debris accumulation · silted water	· moderate · slight
Etobicoke	· algal blooms · *Cladophora* sp. · odour	
Peel	· algae · dead fish · odour	· moderate
Halton	· debris · dead fish · algal blooms · odour	· severe
Hamilton-Wentworth	· creeks entering lake cause most pollution · dead fish	· moderate
Niagara	· seasonal dead fish · algae	

a. Source: Chong (1984).

problems are algae-related (*Cladophora* sp. and other algal blooms) and dead fish. Particular problems of note appear to be at beaches in the Etobicoke and Halton areas. The aesthetic problems at Etobicoke were reported to be increasing; the problems on the beaches of Halton were rated as severe.

5.11.3 ODOUR PROBLEMS

Human perception of air quality or odour appear to be directly related to seasonal responses rather than actual pollution levels (Smardon, 1989). Summer is the time (a) when more people are out observing and using the lake, and (b) the elevation in water temperature increases the vapour pressures of undesirable substances. Indeed, in the Toronto area most complaints are registered from May through October (RCFTW, 1990). Odours may cause nausea, headaches, loss of sleep, and emotional problems. Judging by the Toronto experience (RCFTW, 1990), most olfactory complaints are linked to air pollutants (e.g., sulphur dioxide, sulphides or organic compounds). No reports other than anecdotal accounts specifically mention Lake Ontario water as the source of unpleasant odours.

5.11.4 PUBLIC PERCEPTION OF AESTHETICS

By its very nature, judgements about aesthetic values are subjective. Speaking generally, what one person may view as aesthetically pleasing or displeasing may not be shared by others. When respondents are asked about their perception of water quality or water pollution, visual evidence is frequently cited (Smardon, 1989). Indeed, it is difficult for people to perceive how an altered landscape or lake will appear or to anticipate its change

over time. A means for understanding how the population at large perceives continuous change is lacking. Insights into local populations' ecological education levels, as well as landscape appreciation, tastes, and values, will aid in the preparation of interpretive programs (Smardon, 1989).

5.11.5 HERITAGE PERSPECTIVES

Heritage preservation helps to ensure that a given waterfront remains unique. This uniqueness contributes to the inherent aesthetic value of a given locale. Human influences have altered much of the Lake Ontario shoreline (RCFTW, 1989a). Networks of highways and railways run along the waterfront's edge (e.g., from Burlington, to Newcastle, Ontario). Stonehooking (the collection of rock for building) and dredging have altered the underwater contours of the lake (Whillans, 1979). In addition, lakefilling programs have filled marshes and created harbours/recreation areas.

While there exists some official recognition of the value of heritage preservation and a degree of public support for it, the reality is that the waterfront heritage is not being preserved (RCFTW, 1989a). According to the Crombie Commission, a waterfront-wide heritage preservation policy requires consultation amongst heritage preservation people, the public sector, the private sector, and all levels of government. Currently, the Province has initiated an Ontario Heritage Policy Review to examine activities, legislation, and programs with a view to creating a policy framework within which government programs and legislation might be improved.

Recommendations to help maintain and reinforce the historical continuity of the Toronto, Ontario, waterfront are listed below (RCFTW, 1989b).

1. The preservation of the waterfront heritage should be affirmed as a priority in all plans affecting the waterfront.

2. The concept of heritage should be broadly defined to include, not just architecturally or historically important buildings and archaeological sites, but in addition, districts, landscapes and views, as well as elements of our marine and industrial heritages and historic patterns of development.

3. A sensitive adaptation of historically or architecturally significant buildings should be maintained.

4. There should be preservation of the current shoreline of the central waterfront.

5.11.6 PUBLIC ACCESS

To appreciate the aesthetic quality of Lake Ontario, public access to waterfront areas must be initiated and maintained. This includes both direct physical access to the water's edge and broad economic and social access to waterfront places. A lack of public access to the waterfront is a major concern in the Hamilton Harbour Area of Concern (AOC) (Hamilton Harbour RAP Team, 1989). To maintain and enhance the waterfront's overall diversity, industrial, commercial, residential, recreational, and wilderness uses in development and redevelopment need to be incorporated (RCFTW, 1989b). The Crombie Commission recommends the adoption of ecologically sound planning, development, and use of the land and water at the waterfront. Its recommendations include:

1. the use of uncontaminated landfill only;

2. approval of only those lakefill configurations that do not create environmental damage; and

3. recreational development at the water's edge that does not cause environmental damage;

4. recreational use of the water that does not harm it or the water's edge;

5. facilities and programs that increase awareness of the environment;

6. the practice of good environmental citizenship by the operators of waterfront parks and other public amenities.

In addition, dissemination of information is crucial. People cannot enjoy the many pleasures of the waterfront if they do not know what and where these are, or how to access them.

5.11.7 CONCLUSIONS

It is clear from the data in Table 5.11.1, that algal blooms were a consistent problem in 1984. Since eutrophication is still a problem in the Toronto, Hamilton, Bay of Quinte, and Oswego AOCs, and the expectation of a persistence of current nitrogen and phosphorus levels (Section 5.8), these elements should be designated as causal pollutants.

5.12 ADDED COSTS TO AGRICULTURE OR INDUSTRY

5.12.1 DEFINITION AND DETERMINANTS OF ADDED COSTS TO AGRICULTURE AND INDUSTRY

The twelfth impairment of beneficial uses listed by the International Joint Commission (IJC) constitutes "added costs to agriculture and industry". The following listing and delisting guidelines for this impairment were recently formulated (IJC, 1991).

Listing Guideline

When there are additional costs required to treat the water prior to use for agricultural purposes (i.e., including but not limited to, livestock watering, irrigation or crop-spraying) or industrial purposes (i.e., intended for commercial or industrial applications and non-contact food processing).

Delisting Guideline

When there are no additional costs required to treat the water prior to use for agricultural purposes (i.e., including, but not limited to, livestock watering, irrigation and crop-spraying) and industrial purposes (i.e., intended for commercial or industrial applications and non-contact food processing).

Substantial information is available with respect to the impacts of agriculture and industry on Lake Ontario water quality. However, there is little information available with respect to added costs to agriculture and industry as a result of poor water quality.

5.12.2 ADDED COSTS TO AGRICULTURE

A significant proportion of land use in the Lake Ontario basin is devoted to agriculture. In Ontario, 49% of the basin land area is classified as farmland, compared to 33% in New York State (see Chapter 2).

A number of studies (e.g., Fox and Dickson, 1989, 1990) have been carried out assessing the impacts of agriculture on the Great Lakes, but they are not specific to Lake Ontario. Extra costs are incurred because sediments arising from soil erosion can disrupt fish reproduction and feeding, reduce the quality of recreational activities, decrease the capacity of water storage facilities and navigational channels, increase the frequency and volume of floods, complicate water-treatment and maintenance of water-using machinery and appliances, and clog drainage ditches and irrigation canals (Fox and Dickson, 1990). Research and financial incentive programs such as SWEEP (Soil and Water Environmental Enhancement Project), OSCEPAP II (Ontario Soil Conservation and Environment Protection Assistance Program II), and Tillage 2000 are addressing problems of this type. Hence most of the added costs to agriculture are in the form of remedial or control types of activities.

Some farmers employ water from the Great Lakes for irrigation purposes. However, this practice is so limited that the added costs in terms of using this water cannot be determined (V. Spencer, Ontario Ministry of Agriculture and Food, Guelph, personal communication). It is hard to imagine that under normal conditions Lake Ontario water would ever be unsuitable for irrigation purposes. Of course, contamination of lake water by accidental spills (e.g., chemicals, oil) might well incur special treatment costs. Although there may be some (toxic) accumulation in the soil, the possibility of a negative impact on agriculture is remote. With respect to livestock drinking Lake Ontario water, bioaccumulation would be expected to be minimal as these animals do not consume recognized bioaccumulators such as fish (D. Coote, Agriculture Canada, Ottawa, Ontario, personal communication).

5.12.3 ADDED COSTS TO INDUSTRY

There are occurrences of additional costs required to treat Lake Ontario water prior to use for industrial purposes. Kodak in Rochester, New York, has constructed a water-treatment plant for their own use because of the high water-supply and purity requirements for their production processes. It must be noted that it is possible that water quality may meet the State or Provincial sanitary code or regulations, but may require further purification for complex industrial processes. Another example is Molsons Brewery in Toronto, Ontario, which treats water so as to remove organic compounds (J. Steer, Molson Breweries, Toronto, Ontario, personal communication).

HG Heinz in Leamington, Ontario, was contemplating building a water-treatment plant for Lake Erie water, but the estimated costs proved to be too high; the rationale for this development was economic and not water quality (P. Boyer, International Joint Commission, Windsor, Ontario, personal communication).

Industrial facilities utilizing Lake Ontario as a source for cooling or process water have recently reported zebra mussel colonization of intake pipes. Preventive measures currently being researched to temporarily alleviate the clogging of these intakes include the use of chemical oxidants, such as ozone or chlorine, and mechanical/physical methods such as protective coatings, acoustics, UV and gamma radiation, elevated pressure and temperature, and mechanical filtration. At the moment, most of Ontario Hydro's stations chlorinate intake water from early summer to late fall. However, concerns exist about the environmental impact of chemical measures such as chlorination in generating contaminants (e.g., trihalomethanes) or producing oxidative damage to other aquatic life. Ontario Hydro has

recently estimated an incurred one-time cost of around 3.5 million dollars, with ongoing costs projected at 0.7-1.1 million dollars per year (R. Claudi, Ontario Hydro, Toronto, personal communication).

From this brief review, it is clear that very little information is available on expenditures required to remove pollutants from Lake Ontario water prior to agricultural or industrial uses. A representative from the Canadian Manufacturing Association stated that this dimension is yet to be evaluated (D. Henley, Canadian Manufacturers Association, Toronto, Ontario, personal communication).

5.12.4 CONCLUSIONS

It is apparent that infestation of zebra mussels is the only clear impairment in terms of added costs to agriculture or industry. Since the most common declogging treatment involves the release of chlorine into the water-intake pipe, pollution of lake water with this powerful oxidant may potentially occur. These two reasons might well suffice to designate zebra mussels as a causal agent.

CHAPTER 5.13 DEGRADATION OF PHYTOPLANKTON AND ZOOPLANKTON POPULATIONS

5.13.1 DEFINITION

The International Joint Commission's (IJC) guidelines for impairment 13, "degradation of phytoplankton and zooplankton populations" are shown below.

Listing Guideline

When phytoplankton or zooplankton community structure significantly diverges from unimpacted control sites of comparable physical and chemical characteristics. In addition, this use will be considered impaired when relevant, field validated, phytoplankton or zooplankton bioassays (e.g., Ceriodaphnia; algal fractionation bioassays) with appropriate quality assurance/ quality controls confirm toxicity in ambient waters.

Delisting Guideline

When phytoplankton or zooplankton community structure does not significantly diverge from unimpacted control sites of comparable physical and chemical characteristics. Further, in the absence of community structure data, this use will be considered restored when phytoplankton or zooplankton bioassays confirm no significant toxicity in ambient waters.

The rationale given for these guidelines is that they account for community structure and composition, recognizing water column toxicity, and use an appropriate control site (IJC, 1991). The guidelines have two distinct components; one is based on the ecological concept of community structure; and the other is based on a statistically derived toxicity test. The guideline is very similar in this way to the guidelines for impairment 6, the degradation of benthos. Several important terms such as "community structure", "significantly", "relevant" or "appropriate" are not defined. This leaves the interpretation of the data, and whether the plankton meets the guidelines, up to each agency.

This section reviews the available data on phytoplankton and zooplankton community structure and bioassays. The phytoplankton discussion is closely linked to Section 5.8, eutrophication or undesirable algae.

5.13.2 DEGRADATION OF PHYTOPLANKTON POPULATIONS

Phytoplankton populations are one of the first steps in the food chain in an aquatic ecosystem. Phytoplankton can bioaccumulate and biomagnify contaminants and so play an essential role in contaminant cycling. Phytoplankton populations are influenced by a number of biotic and abiotic factors including phosphate concentrations, competition, predation, light, temperature, and wind. Phytoplankton populations have two major aspects: species composition and abundance. Both of these aspects show a distinct seasonal cycle, with some species dominating in spring and others in fall. In an oligotrophic lake, phytoplankton tend to exhibit several minor peaks throughout the year, whereas in a meso-eutrophic system they often have a bimodal pattern with a midsummer minimum. In eutrophic conditions, the larger net plankton tend to dominate the biomass, whereas in oligotrophic or mesotrophic conditions the smaller nannoplankton dominate (Stevens, 1988).

In determining long term trends in phytoplankton populations, three methods are generally used: seasonal changes in biomass; changes in the relative contribution of algal species to the total biomass; and changes in species which are taken to be indicators of trophic status. As there can be significant changes in all three of these measurements from one year to the next, long-term data sets with consistent sampling, storage, and analysis methods are required in order to assess trends. A major handicap to the interpretation of

long- term trends in Lake Ontario is the lack of standardized methods of sampling, storage, and analysis. Sampling methods employing an integrated sampler may yield different results than sampling at discrete depths. Species identification can vary among workers, resulting in inconsistencies on species composition for the lake. Quantitative methods for algal biomass and species composition can give different results. The importance of standardization of methods has been emphasized by the IJC (1989).

Several major lake-wide studies of phytoplankton populations have been conducted on Lake Ontario. The earliest recorded studies date from the late 1960s, followed by a studies in 1970, 1972, 1975, 1977, and 1978 (Stevens, 1988). In 1981-1982, two major studies were conducted on Lake Ontario. One, the Bioindex program selected three sites representing nearshore and open water situations, which were monitored weekly. The second program, Lake Ontario Nutrient Assessment Study, LONAS, sampled phytoplankton monthly along a north-south transect. The different results found in these two studies illustrate the differences which can occur from different sampling methods and counting. Some 150 species of phytoplankton were identified in the Bioindex program, with 40 of these species considered common and contributing 5% of the biomass. In the LONAS program, 174 species were identified with 49 of them considered common. However, only 21 of the species were considered common in both studies. Using a weighted median, the LONAS program calculated seasonally weighted mean algal biomass of 0.43 g/m^3 in the midlake in 1982; using a weekly weighted average, the Bioindex program reported more than double that biomass, 0.93 g/m^3, (Johannsson et al., 1985; Stevens, 1988).

5.13.2.1. Phytoplankton biomass. The most distinctive change in phytoplankton population biomass between 1970 and 1982 was a shift from a unimodal pattern with a large summer maximum to a bimodal pattern with a summer minimum. Despite this shift in population dynamics, there has been little shift in the dominance of the common species since 1967 (Stevens, 1988).

Generally, phytoplankton biomass is greater in the nearshore areas compared to the midlake (Gray, 1987; Johannsson et al., 1985). This difference in nearshore and offshore phytoplankton biomass has also been observed in Lake Erie and Lake Michigan, whereas the upper Great Lakes have no well developed nearshore-offshore gradients (Munawar and Munawar, 1981). These nearshore-offshore gradients are promoted by physical events such as thermal bars, boundary currents, wind stress, and storms (Gray, 1987).

In 1982, phytoplankton biomass was characterised by multiple peaks which ranged over three orders of magnititude. Mean biomass was 43 to 58% less in 1982 compared to 1972 (Gray, 1987). However, sampling differences exist between these two time periods. In 1972, biomass samples were collected using an integrated sampler, and in 1982 samples were collected at 5 m as this depth was considered representative of the upper mixing zone. These sampling differences were considered unimportant by some workers (Gray, 1987), but sufficient to permit only cautious interpretation of data by other authors (Johannsson et al., 1985). Another complicating factor in comparisons over time is the role of heterotrophic bacteria and microflagellates. Much of the particulate biomass can consist of heterotrophic bacteria and heterotrophic microflagellates which were generally not enumerated prior to 1982 (McLean et al., 1987).

Mean summer algal biomass in Lake Ontario was twice as high in 1970 as in subsequent studies at all nearshore and offshore stations in the LONAS and Bioindex programs. The 1975 survey shows slightly higher but probably statistically insignificant biomass than in the 1977 and 1982 surveys. There appears to be no distinct trend in algal biomass since 1974 (Stevens, 1988).

Chlorophyll a did not show a similar reduction over time. Chlorophyll a content per cell volume can vary markedly between species and so may account for the lack of relationship between biomass and chlorophyll a.

5.13.2.2. Species Composition. In 1965, phytoplankton at 11 stations in northeastern Lake Ontario had a spring-summer maximum of 3.1×10^6 to 60.8×10^6 μ^3/mL, where μ^3 is the volume of microalgae. Diatoms accounted for 53%, flagellates for 28%, Chlorophyta for 13% and Cyanophyta for 13% of the annual total phytoplankton volume (Sreenivasa and Nalewajko, 1975). This would indicate that in 1965, the northeastern basin of Lake Ontario was highly eutrophic, as the maximum plankton volume exceeds the cut-off point of 10×10^6 μ^3/L suggested by Vollenweider (1968).

In 1981 and 1982, 150 species of phytoplankton were identified in Lake Ontario. The predominant groups were Bacillariophyceae (diatoms), Cryptophyceae, and Dinophyceae. The fluctuations in algal biomass over the season are mainly a result of fluctuations in these three groups. The percent contribution of algal species remained relatively constant among stations and between the years 1981 and 1982 (Johannsson et al., 1985). Diatoms dominated the phytoplankton population in 1982 in the nearshore zone, whereas greens and phytoflagellates dominated the midlake station (Gray, 1987).

5.13.2.3. Changes in Phytoplankton Species. In midlake, the mean seasonal proportion of nannoplankton to total phytoplankton did not change between 1972 and 1982, whereas net plankton was less frequent in 1982 (Munawar et al., 1984). There has also been a shift in dominant summer phytoplankton from blue-greens and greens in 1970 and phytoflagellates in 1972 to greens and phytoflagellates in 1982 (Gray, 1987).

There has been a decline in the number of species considered to be indicative of eutrophic conditions (Gray, 1987). Certain species of Bacillariophyceae are often used to indicate trophic status of areas. *Melosira binderana, Stephanodiscus tenuis, S. hantzschii var pusilla*, and *S. alpinus* are all classified as meso-eutrophic indicators. These species were found in large numbers in the nearshore zone during the winter to spring of 1970, and only in small numbers in the midlake stations (Munawar and Nauwerck, 1971). These species were missing from the nearshore and midlake samples taken in 1982 (Gray, 1987). *Tabellaria flocculosa* and *Melosira islandica* are considered oligotrophic species, and were found in both the nearshore and midlake stations in 1982 (Gray, 1987). Another study using phytoplankton community structure also shows that the flora of Lake Ontario was more eutrophic in the late 1960s and early 1970s. In 1966 -1968, three species characteristic of eutrophic conditions were found, which increased to seven in 1970, and decreased again to four species in 1981-1982 (Johannsson et al., 1985). Changes in species composition can be a result of bottom up factors such as nutrient changes and/or top down factors such as zooplankton grazing.

5.13.2.4. Toxicity. Certain species and sizes of phytoplankton are more sensitive to toxic contaminants than others. This difference in suspectibility has two implications: (1) the more sensitive species can be used for bioassay screening tests; (2) and the more sensitive species will tend to be absent in conditions of high contamination in Lake Ontario. Generally, the smaller sizes of phytoplankton such as picoplankton (<2.0 μm) are the most sensitive to contamination, especially metal mixtures (Munawar et al., 1987). This has major ecological consequences, as the ultraplankton/picoplankton (<20 μm) dominate the phytoplankton of most of the Great Lakes and provide the major diet for zooplankton and fish. Toxic substances can cause the reduction or elimination of plankton Of these sizes, and so alter the food chain in the Lakes.

In 1988 in Ashbridges Bay near Toronto, concentrations of bacteria, heterotrophic flagellates and micro- and netplankton were not significantly different than a Lake Ontario transect. However, the concentration of autrophic picoplankton was significantly lower in Ashbridges Bay and Hamilton Harbour compared to the Lake Ontario transect (Munawar and Weise, 1989). Autrophic picoplankton appear to be extremely sensitive to toxic pollution. This is in agreement with Munawar and Munawar (1987), who compared the abundance of Niagara River picoplankton with that of Lake Ontario and concluded that their scarcity in the river was due to metal contamination.

A size-fractionated algal bioassay was performed on Ashbridges Bay phytoplankton in 1988. Elutriates from sediment in Ashbridges Bay were toxic to both ultra/picoplankton (<20 μm) and micro/net plankton (>20 μm) from offshore Lake Ontario at the EC_{50} of 15% and to *Dapnia magna* (Munawar et al., 1989).

Some analysis has been completed on contaminant concentrations in phytoplankton, most particularly in *Cladophora*. *Cladophora*, a filamentous green alga, is used as a biomonitor of contaminants in the Lower Great Lakes by the Ontario Ministry of Environment, and also has been used by the New York State Department of Environmental Conservation. While the definition of impairment does not include the use of contaminant concentrations in phytoplankton, there is a comprehensive database which clearly pinpoints specific areas of contamination, such as PCBs and mercury along the Niagara River and PCBs in the Humber and Credit Rivers. The effect on the physiology of *Cladophora* from these elevated concentrations is not well known. A consistent decrease in PCB, DDE, and dieldrin has been observed in netplankton in Lake Ontario from 1972 to 1982 (Government of Canada, 1991a).

Degraded phytoplankton populations are considered an impaired use in three of the eight Areas of Concern on Lake Ontario; Niagara River, Hamilton Harbour, and Bay of Quinte. In Oswego, phytoplankton studies in 1981 indicated that the harbour and river area were eutrophic. Algal species typical of halotrophic (salty) conditions were also observed and related to the salt storage piles in the harbour area (Bertram, 1985). In 1986, improvements were made to the sewer discharges, which were expected to reduce nutrient and toxic loadings. In 1987, the salt storage was discontinued. The RAP concludes that while this beneficial use had been impaired in the past, there is no recent phytoplankton or zooplankton data to judge the current state of impairment (NYDEC, 1990). The RAs for Eighteenmile Creek and Rochester are currently evaluating this impairment.

In the Niagara River, major differences in species composition occur in the algal size class 5 - 20 μm compared to the midlake. The Niagara River also had lower biomass than Lake Ontario (Niagara River RAP Team, 1990). In the Toronto RAP, the report concludes that there are insufficient data to determine the significance of local sources versus lake-wide sources on degradation. Phytoplankton collected weekly at a Toronto filtration station had shown a doubling in annual algal levels from 1923 to 1963. There was also a shift in dominance from *Asterionella* to *Cyclotella* after 1938 (Metro Toronto RAP Team, 1989).

In the Hamilton Harbour, phytoplankton taken from the Harbour were inhibited by mixtures of metals at the minimal concentrations specified by Great Lakes Water Quality Objectives (Wong et al., 1978). However, the average concentrations of metals in Hamilton Harbour in 1984-1987 were generally below these objectives, with the exception of zinc which exceeded the objectives 3% of the time in 1986-1987 and iron 9% of the time. This represents a major decrease in metal concentrations since 1982 when copper exceeded the guidelines 69% of the time, zinc 30% and iron 33%. Therefore, while Wong's study illustrates the synergistic effects of the metals at GLWQA objective concentrations, these concentrations are not currently found in the waters of Hamilton Harbour. Similarly, copper and mercury were found to inhibit photosynthesis in the Harbour in 1982, but metal concentrations have decreased since this study (Hamilton Harbour RAP Team, 1990).

In the Port Hope RAP, "fish population densities and community structure in the harbour and surrounding area indicate a viable plankton community". There are no data presented to support this conclusion, zooplankton degradation is not considered and few data are presented to support the conclusions on fish populations. The dismissal of degraded phytoplankton and zooplankton populations in the Port Hope RAP seems premature. In the

Bay of Quinte, there has been a four fold reduction in the total algal biomass over 16 years, but the species composition remains much the same; diatoms and blue-greens account for 85% of the biomass (Bay of Quinte RAP Team, 1990).

Therefore, while many of the RAP areas have documented degraded phytoplankton populations in the past, in the current situation, there are either not enough recent data to assess this impairment or those areas do not seem to be impaired.

5.13.3 ZOOPLANKTON POPULATIONS

Zooplankton populations provide the integration between "top down" and "bottom up" effects, as they prey on phytoplankton populations and are preyed upon by fish populations. The most important groups in Lake Ontario are Rotifera, Crustaceae and Protista. Crustacean zooplankton are divided into three types: cyclopoid and calanoid copepods, cladocerans, and Malacostraca. *Mysis relicta, Daphnia sp., Bosmina sp.,* and *Cyclops* are some of the more common zooplankton in Lake Ontario.

Many factors contribute to the community structure of zooplankton. Physical factors such as water temperature, turbulence, upwellings, light and substrate, and biological factors such as fish community structure, phytoplankton community structure will all interact to form zooplankton community structure. Because of these numerous factors, all of which are dynamic and themselves dependent on other multiple factors, teasing out the influence of chemical contamination on community structure is a difficult task. In addition, assessing the degradation in zooplankton populations is subject to similar caveats as phytoplankton populations. Differences in sampling depth, frequency, time, and net size can significantly alter results from one study to another. For example, studies using different vertical depth

of sampling will produce different results as each zooplankton species has a particular depth of feeding which can vary over the day or season (Johannsson et al., 1985).

5.13.3.1. Zooplankton Biomass and Species Composition. Zooplankton biomass is generally higher in nearshore areas compared to midlake stations (Johannsson et al., 1985). The community structure of zooplankton in the sense of species present has not significantly changed from 1967 to 1982, but the relative abundance of species has changed. There have been increases in cyclopoid copepods and *Bosmina longirostris* (Johannsson et al., 1985). In a more recent followup study comparing changes in nearshore and offshore zooplankton abundance and composition between 1981 and 1988 with historical data, no change had occurred in the basic community structure since the late 1960s (Johannsson et al., 1991). Abundance had decreased in the midlake and nearshore zone from 1983/1984 to 1987. The nearshore zone had more small cladocerans and greater year-to-year variability; the midlake zone had more of the larger cladocerans (Johannsson et al., 1991).

In the RAP areas on Lake Ontario, zooplankton populations are considered an impaired use in Niagara, Hamilton Harbour, and Bay of Quinte, are under evaluation in 18 Mile Creek and Rochester; and they are not considered an impaired use in Port Hope, Toronto or Oswego. In Port Hope, no data are presented to support their decision. In Toronto, shifts in community structure were observed between 1939 and 1972. Recent data indicates that thermal upwellings may play a larger role in determining community structure than first believed, and that this may significantly alter zooplankton in the nearshore, north shore of Lake Ontario. The RAP concludes that not enough data exist to determine the impact of local sources on zooplankton compared to lake populations (Metro Toronto RAP Team, 1990). In Hamilton Harbour, the zooplankton is typical of eutrophic conditions, with

large rotifers dominant. Few species of zooplankton can survive in the anoxic bottom layers of Hamilton Harbour, and this limits the zooplankton population. In an early study, water from Hamilton Harbour was not found to be toxic for zooplankton, and this is under review (Hamilton Harbour RAP Team, 1989). In the Bay of Quinte, cladocerans have decreased in abundance from the upper to the lower bay and cyclopoids increased.

5.13.4 CONCLUSIONS

Phytoplankton biomass increased in the late 1960s and early 1970s, decreased in the mid-1970s and has remained relatively constant in the 1980s. Phytoplankton community structure shifted from oligotrophic species to eutrophic species in the 1960s and 1970s and in the early 1980s is shifting back from these eutrophic species. The biomass and species composition of phytoplankton varies from nearshore to offshore areas in Lake Ontario. In several of the nearshore areas, impairment of phytoplankton populations is apparent.

Zooplankton populations also show nearshore-offshore gradients. Community structure in the early 1980s was similar to the 1960s and early 1970s. There is limited evidence of degradation of zooplankton populations.

Most of the studies on zooplankton and phytoplankton populations have focused on the ecological aspects of community structure and not correlated this with contaminant concentrations. Bioassays of phytoplankton and zooplankton often give conflicting results. More work is needed on this aspect before causal pollutants can be identified with confidence.

5.14 SUMMARY OF IMPAIRED BENEFICIAL USES AND CAUSAL POLLUTANTS / AGENTS IN LAKE ONTARIO

5.14.1 LOCATIONS OF IMPAIRED BENEFICIAL USES

All locations in Lake Ontario where an impairment was documented in this inventory are shown in Table 5.14.1. In certain cases, regions (e.g., Hamilton-Wentworth, Halton, Peel, Durham) are listed because more specific locations were not available. This summary is probably incomplete with respect to New York waters, pending the completion of a similar study by the U.S. Environmental Protection Agency and the State of New York.

5.14.2 CAUSAL POLLUTANTS / AGENTS

Based upon information documented in the inventory of impairment of beneficial uses, a list of causal pollutants/agents was compiled (Table 5.14.2). The criterion that we used to generate the list of causal pollutants / agents was a demonstrated or strongly suspected association with a beneficial use impairment (by weight-of-evidence) on the condition that there was some evidence of the presence (analytical measurements showing elevated levels) of that pollutant/agent in Lake Ontario, i.e., a record of the pollutant being present in at least one of sediment, water, or biota. For each beneficial use impairment, a review of the literature using the critical appraisal approach identified whether an association between a pollutant/agent and an impairment existed. When the evidence of association was weak or suspected it was noted in the list.

Table 5.14.1. Lake Ontario locations where impaired beneficial uses have been documented, 1970-1991.

	Beneficial Use Impairment												
Location	1	2	3	4	5	6	7	8	9	10	11	12	13
Ontario													
Niagara River	X	U	X	X	U	X	X	N	U	U	U	U	X
Niagara (specifically St. Catharines)[a]										X	X		
Jordan Harbour	X												
Grimsby							X						
Hamilton-Wentworth[a]										X	X		
Hamilton Harbour	X	U	X	X	X	X	X	X	N	X	X	X	E
Halton[a]										X	X		
Bronte Creek	X												
Oakville Creek				X									
Peel[a]										X	X		
Credit River	X						X						
Metro Toronto	X	D	X	O	O	X	X	X	U	X	X	U	D
Durham[a]										X	X		
Frenchman's Bay	X												
Pickering NGS	X												
Whitby					X		X						
Oshawa	X						X						
Port Hope	X	U	D	X	D	X	X	X	N	N	U	N	D
Haliburton, Kawartha and Pine Ridge[a]										X			
Presqu'ile Bay	X			O									
Hastings and Prince Edward[a]										X			
Scotch Bonnet	X				X								
Gravelly Bay	X												
Bay of Quinte	X	U	X	X	D	X	X	X	X	X	X	U	X
Prince Edward Bay	X												
North Channel	X												
Gull Island					X								
Pigeon Island			X		X								
Main Duck Island	X												
Snake Island			X		X								
Brother Island			O		O								
Delta			O		O								
Lower Gap	X												
Reeds Bay	X												
Point Traverse							X						

Table 5.14.1. Continued.

Location	Beneficial Use Impairment												
	1	2	3	4	5	6	7	8	9	10	11	12	13
New York													
Niagara River	E	E	E	E	E	E	E	E	E	E	E	E	E
Olcott							X[b]						
Eighteenmile Creek	X	E	E	E	E	E	E	E	E	E	E	E	E
Twelve Mile Creek	X												
Oak Orchard	X												
Rochester	X	E	E	E	E	E	X[b]	E	E	E	E	E	E
Oswego	X	D	X	D	D	D	X[b]	X	N	N	D	U	D
Sodus Bay	X												
Salmon River	X												
Sandy Creek	X												
Pultneyville	X												
Hardscrabble	X												
Stony Island	X												
Black River Bay	X												
Chaumont Bay	X												
Henderson Bay	X												
Eastern Lake Ontario					X								
New York shoreline										X[a]			

[a] Specific locations on Lake Ontario within the region were not stated in the documents reviewed.
[b] The Oswego RAP indicates no restriction, but a comparison of the chemical data with the EPA guidelines indicates disposal could be restricted (Section 5.7).

X - Use currently impaired.
O - Use not currently impaired, but was documented as impaired in the 1970s or early 1980s.
U - No use impairment.
E - Use impairment currently being evaluated.
D - No data with which to judge use status.
N - Use impairment not relevant.

Beneficial use impairments are as follows:

(1) restrictions on fish and wildlife consumption
(2) tainting of fish and wildlife flavor
(3) degradation of fish wildlife populations
(4) fish tumors or other deformities
(5) bird or animal deformities or reproduction problems
(6) degradation of benthos
(7) restrictions on dredging activities
(8) eutrophication or undesirable algae
(9) restrictions on drinking water consumption, or taste and odour problems
(10) beach closings
(11) degradation of aesthetics
(12) added costs to agriculture or industry
(13) degradation of phytoplankton and zooplankton populations

Table 5.14.2. List of Causal Pollutants / Agents identified in Lake Ontario.

Pollutant / Agent	Beneficial Use Impairment												
	1	2	3	4	5	6	7	8	9	10	11	12	13
Organic and Inorganic Pollutants													
arsenic							X						
cadmium							X						
chromium							X						
copper							X						
DDT/DDE			X		X								
dioxins/2,3,7,8 TCDD	X		X[a]		X								
lead							X						
mercury	X												
mirex	X												
nickel							X						
nitrogen								X	X		X		
PAHs				X		X							
PCBs	X		X[a]		X		X						
phosphorus							X[a,b]	X			X		
zinc							X						
TCDF						X							
Surrogate Measures													
chemical oxygen demand							X						
oil and grease							X[a]						
TKN							X[a]						
volatile solids							X[a]						
Biological Agents													
bacteria										X			
zebra mussels												X	

[a] Denotes a suspected association.
[b] Total phosphorus.

The criteria that we used are similar to the criteria used in the Lake Ontario Toxics Management Plan (LOTMP) and the Great Lakes Water Quality Agreement (GLWQA). The LOTMP used presence in Lake Ontario in ambient water column and/or fish data and exceedance of standards, criteria, or guidelines. Since many of the impaired beneficial uses catalogued in our study are based on exceedances of standards and guidelines, our selection

criteria are similar to those used by the LOTMP through the Lake Ontario Categorization Committee. The GLWQA (supplement to Annex 1) specifies three lists, and our criteria correspond most closely to the criteria for List No. 1 chemicals:

> *List No. 1 shall consist of all substances (1) believed to be present within the water, sediment or aquatic biota of the Great Lakes System and (2) believed, singly or in synergistic or additive combination with another substance, to have acute or chronic toxic effects on aquatic, animal or human life.*

5.14.3 EXISTING POLLUTANT LISTS

Numerous organizations have developed pollutant lists, some of which are summarized in this section (Table 5.14.3). Although the lists themselves are often similar with respect to pollutants listed, the selection criteria or approach used often vary. The criteria most often used involve a combination of toxicity evaluations and hazard assessments. The National Wildlife Federation/Canadian Institute for Environmental Law and Policy, (NWF/CIELAP) (1991) list is an exception since it is based solely on the bioconcentration potential of the pollutants.

There are several generic limitations to the use of pollutant lists, including the list generated by this report, that bear mentioning. The creation of a list tends to focus attention on the appearance and placement of individual contaminants on that list rather than the damage that a contaminant may cause in the field. Furthermore, the fact that a contaminant appears on a list is frequently cited as a rationale for studying or monitoring that contaminant in the environment. These types of activity, out of context of the list, bias the perceived utility of that list. Lists of pollutants only serve their intended purpose if they are updated as new information becomes available, but the resources to pursue such activities are rarely

Table 5.14.3. A Comparison of selected pollutant lists relevant to Lake Ontario.

Organization	Name of List	Selection criteria or Approach
Binational Objectives Development Committee (BODC, 1989)	(1) Present and Toxic (2) Present and Potentially Toxic (3) Potentially Present & Toxic	•Toxicity evaluation •Presence in Great Lakes
Canadian Environmental Protection Act (CEPA)	Priority Substances	
Great Lakes Water Quality Agreement (GLWQA, 1989)	Hazardous Polluting Substances (Annex 10)	•Toxicity evaluation •Discharge potential
Great Lakes Water Quality Board (GLWQB, 1985)	Persistent Toxic Substances (Annex 1)	•Presence in Great Lakes •Persistent, toxic, and bioaccumulative
Lake Ontario Toxics Committee (LOTC, 1989)	(1) Category I Toxics (2) Category II Toxics	•Availability of ambient data •Level or input to Great Lakes
Michigan Department of Natural Resources (MDNR, 1980)	Critical Materials Register	•Recognized high toxicity •Selection from other lists •Specific to L. Michigan •Assessment methodologies •Numerical scoring system
Ontario Ministry of the Environment (OME, 1988)	Effluent Monitoring Priority Pollutants	•Chemical identification •Preliminary assessment
Ontario Ministry of the Environment (Cantox, SENES and OME, 1986)	Chemical Contaminant Priority List	•Environmental behavior •Toxicity evaluation •Level of Concern scoring
National Wildlife Federation Canadian Institute for Environmental Law and Policy (NWF/CIELAP, 1991)	Priority Pollutants	•Bioconcentration
Niagra River Toxics Committee (NRTC, 1984)	(1) Group I (2) Group II (3) Group III	•Criteria information •Chemical/toxicological •Environmental occurence
U.S. Environmental Protection Agency (see NRTC, 1984)	Priority Pollutants	•US court approved EPA Committee "wish list" of compounds to be analyzed
U.S. Environmental Protection Agency	High Priority Pollutant List	

provided in a timely fashion. Most lists are compiled as an expression of agency concern about the regulation of chemicals. However, there is a lack of data with respect to chemical composition of commercial products, which creates difficulty in identifying those products which may contain toxic substances.

Our list of causal pollutant/agents is based on elevated levels in Lake Ontario and association with an impaired beneficial use(s). The lists summarized in Table 5.14.3 used criteria with varying degrees of similarity. We present this comparison only to show that most lists, including ours, contain the same contaminants. Where there are differences they are attributable to the selection criteria.

In three of the pollutant lists examined (BODC, 1989; LOTC, 1989; and NRTC, 1984) multiple lists were given. In these cases there may be discrepancies among the lists themselves, with some pollutant(s) appearing on more than one list. To avoid this potential confusion, we presented only one list of causal pollutants/agents (Table 5.14.2).

CHAPTER 6 DISCUSSION

6.1 CONNECTIONS AMONG IMPAIRMENTS

Beneficial uses are impaired in at least 35 locations in Ontario and at least 17 locations in New York (Table 5.14.1) in addition to those pertaining to the eight inshore Areas of Concern. The most frequently documented impairment concerned restrictions on fish and wildlife consumption, followed by restrictions on dredging activities, degradation of aesthetics, and beach closings. The majority of these impairments relate to existing jurisdictional standards (see Section 6.2); if a standard has been violated, then an impairment has occurred. Both New York and Ontario have devoted considerable resources to their respective monitoring programs for advisories on human consumption of contaminated fish, hence this issue is now well documented. Some impairments such as beach closings and degradation of aesthetics seem to occur together; some beach closings may be related directly to degraded aesthetics (Sections 5.10 and 5.11). For other impairments, such as tainting of fish and wildlife flavour and added costs to agriculture and industry, we found little or no information relating to Lake Ontario.

The Lake Ontario Areas of Concern (AOCs) for which Stage I Remedial Action Plan (RAP) reports have been submitted (Niagara River (Ontario), Hamilton, Toronto, Port Hope, and Oswego Harbours, and Bay of Quinte) are the locations with the most documented impairments of beneficial uses. The considerable attention and resources which the RAP process has focused on the AOCs creates the impression of localization of impairments to

these areas. This is probably a misconception. For example, fish populations are generally considered to be degraded relative to conditions in the late 1700s, but with the exception of nearshore fish communities in the AOCs, fisheries management priorities focus predominately on the offshore community. Degradation of nearshore fish communites in locales other than the AOCs is thus poorly documented.

We note some caveats concerning the interpretation of our findings from a review of impaired beneficial uses in Lake Ontario as summarized in Table 5.14.1. The principal problem is that the documented occurrence of an impaired beneficial use depends in part on the relevant research effort. Impairments such as degradation of aesthetics, tainting of flavor, degradation of benthos, and added costs to agriculture and industry have not been seriously studied by researchers in the context of the Great Lakes Water Quality Agreement (GLWQA). There is a substantial database for phosphorus loadings and eutrophication, but much of it relates to offshore conditions and only one pollutant, phosphorus, is identified; such information does not fit well into the location-specific summary shown in Table 5.14.1. Lastly, although we are confident that we assessed most of the major databases and relevant literature and have consulted the appropriate experts, our study is incomplete with respect to New York waters pending the completion of a similar study by the U.S. Environmental Protection Agency and the State of New York. We consider it unlikely that additional information from New York waters will lead to major changes in our conclusions.

We are confident that our findings with respect to consumption advisories, restrictions on dredging activities, and restrictions on drinking water consumption represent the state of

Lake Ontario reasonably well. Both Ontario and New York have relatively large programs to monitor these impairments, probably because they represent direct routes of human exposure to toxic substances.

Our assessment of thirteen impairments of beneficial uses in Lake Ontario generated a list consisting of 22 causal pollutants, causal agents, and surrogates (Table 5.14.2). Causal agents are organisms that by their presence or activities impair a beneficial use (e.g., bacteria as in Section 5.10 and zebra mussels as in Section 5.12). Parameters such as COD, TKN, and volatile solids (loss on ignition) are not causal pollutants on their own, but are surrogate measures of variables that are difficult or impossible to measure on a routine basis and are usually used in specific circumstances. For example, volatile solids is a measure of the organic content of sediment or water and as such includes organic matter from organisms and organic chemicals plus water content. Chemical oxygen demand (COD) is a measure of the organic matter content of a sample that is susceptible to oxidation by a strong chemical oxidant and can be related empirically to BOD, organic carbon or organic matter (American Public Health Association (APHA) et al., 1985). Total Kjeldahl nitrogen, TKN, is a measure of the organic nitrogen content of a sample, including ammonia nitrogen (APHA et al., 1985).

The list of causal pollutants and agents should be used with caution as it may reflect the perspective of the monitoring programs whose data we accessed. For example, pesticides (persistent toxic substances in the GLWQA) such as lindane, chlordane, and aldrin/dieldrin are almost completely absent from our list, although they appear on many agency lists of chemicals that are of concern. There are several possible reasons for the omission of these chemicals: (1) they are present but do not exceed enforceable standards (e.g., see Section 5.1

on fish consumption restrictions); (2) they are not considered a problem and so attention has been focused on other chemicals about which more is known (e.g., PCBs; see Section 5.5); (3) there are no standards or guidelines for these compounds, e.g., the former OWDGs in Ontario considered heavy metals and PCBs only whereas the new PSQGs include a long list of guidelines for organic substances including PCBs, organochlorines, and PAHs (see Section 5.7); or (4) they co-occur with other chemicals such as PCBs, dioxins and furans and it is impossible to distinguish the separate effects of individual contaminants, e.g., with respect to bird deformities and reproductive problems (Section 5.5). Other pollutants such as cadmium, lead, and nickel, and surrogate measures are primarily associated with dredging restrictions (Section 5.7) and likely reflect earlier priorities as new guidelines emphasizing organic substances have been formulated. Lastly, different criteria have been used in practice to judge causal pollutants/agents because of the differences between ecological and jurisdictional impairments (see Section 6.2 below). We used our weight-of-evidence approach (Chapter 4) primarily in the ecological context.

Several contaminants were cited as the causal pollutants of more than one impaired beneficial use in Lake Ontario (Table 5.14.2). The organochlorines, particularly PCBs, dioxins, DDT/DDE, and PAHs, were notable in this respect. We are confident that the organic compounds in our list - PCBs, PAHs, dioxins, DDT/DDE, and mirex - do cause geographically widespread ecological and jurisdictional impairments and should be designated as lake-wide causal pollutants. We are less confident in listing some of the heavy metals and surrogate measures because of the concerns described above.

6.2 DISCUSSION OF IMPAIRMENT LIST

There are two general classes of impaired beneficial uses as defined by the International Joint Commission's (IJC) listing/delisting criteria: (1) those for which "informal" semi-quantitative ecological criteria are used to delineate the impairment; and (2) those for which "formal" quantitative legal or jurisdictional standards, guidelines, or criteria are used to determine when an impairment has occurred. Degradation of fish and wildlife (Section 5.3), and of phytoplankton and zooplankton populations (Section 5.13), fish tumors and deformities (Section 5.4), bird and animal reproductive problems (Section 5.5), degradation of benthos (Section 5.6), and eutrophication (Section 5.8) are examples of ecologically based impairments. Restrictions on fish and wildlife consumption (Section 5.1), on dredging activities (Section 5.7), on drinking water consumption (Section 5.9), and beach closings (Section 5.10) are jurisdictional impairments. For the most part the latter are impairments in which human exposure to toxic substances may occur directly. The standards and guidelines used are therefore set to protect human health, although they may also protect the health of aquatic ecosystems.

It is much more difficult to establish or demonstrate a cause-effect linkage between human interventions and ecological impairments than it is between human activities and jurisdictional impairments, as highlighted in Table 5.14.1. There is a considerable body of scientific knowledge underlying the formulation of legal criteria whereas this is not necessarily so with respect to ecological criteria. In our assessment we noted where legally enforceable standards or guidelines were exceeded as impairments of beneficial uses. We did not attempt to second-guess the information used to set legal standards, guidelines or criteria.

In contrast, a definitive case, in the sense of logical rigorous proof, is impossible to provide in an ecological context because of the limitations of the available data. An ideal of rigorous proof can be approached more closely with abiotic physical phenomena than with biotic ecological phenomena.

We developed and used critical appraisal guidelines to assess the reliability of evidence that we reviewed (Chapter 4) and so constructed weight-of-evidence arguments linking causes and effects. This approach here relies on informed judgement and rests on the weight of evidence built up through applying rules of evidence objectively or in a fair pluralistic context to formulate causal inferences. This approach is not new, but it is only in recent years that it has begun to be applied explicitly to the Great Lakes (e.g., Colborn et al., 1990; Gilbertson and Schneider, 1991).

According to Annex 2 of the Great Lakes Water Quality Agreement (GLWQA) as amended in 1987, Lakewide Management Plans (LAMPs) are required to include "[a] definition of the threat to human health or aquatic life posed by the Critical Pollutants, singly or in combination with another substance, including their contribution to the impairment of beneficial uses". None of the 14 impairments also listed in Annex 2 directly addresses human health issues relating to toxic substances in the Great Lakes. Three of the impairments - restrictions on fish and wildlife consumption (Section 5.1), restrictions on drinking water consumption (Section 5.9), and beach closings (Section 5.10) - indirectly consider human health in that they use standards ostensibly set to protect public health. But so far as we are aware there is no consensual definition of the threat to human health by pollutants in the Great lakes, although the work by the Government of Canada (1991a, b) is

a good first step. Paradoxically, the implications of the threat from toxic substances to wildlife and aquatic life are fairly well known (see Sections 5.3, 5.4, 5.5, 5.6, and 5.13).

The commonalities in our findings, based on the review of beneficial use impairments in Lake Ontario and supporting documentation from the other Great Lakes (see Chapter 5), may be of use in defining the human health threat. We repeatedly came to the conclusion that the weight-of-evidence (as we used the term, see Chapter 4) pointed to the role of chemicals or in most cases, classes of chemicals, as an important determinant of the health of a variety of terrestrial and aquatic organisms with different modes of life. Not only were the same class of chemicals consistently indicted, their effects were also generally consistent between species. For example, the weight-of-evidence strongly supports the conclusion (see National Research Council/Royal Society of Canada (NRC/RSC), 1985; Great Lakes Science Advisory Board, GLSAB, 1989) that reproductive anomalies in lake trout (Section 5.3.1.2), chick edema disease in colonial fish-eating birds (Section 5.5), embryonic deformities and abnormalities in snapping turtles (Section 5.5), and reductions in shoreline mink populations (Section 5.3.2.1) are caused by PCBs. Furthermore, the youngest life stages, zygote and embryo, seem to be most sensitive to the adverse effects of chemicals, whether teratogenic, mutagenic, or carcinogenic, and the least able to defend against chemical intrusions. Adult animals are consistently observed with higher organochlorine concentrations in their tissues than levels reported to cause mortality, teratogenic, or mutagenic effects in embryos.

Several RAPs for Lake Ontario AOCs have modified the list of impairments. In the Bay of Quinte four of the original 14 impaired uses have been dropped because they were not considered to be a concern (Bay of Quinte RAP Team, 1990). The remaining 10

impaired uses were used to compile a revised list of nine impairments specific to the Bay of Quinte, of which six were related to elevated nutrient levels and three were associated with increased levels of toxic substances and bacterial contamination. The Niagara River (Ontario) RAP devised a list of 23 environmental concerns, of which 17 were considered impairments specific to the AOC. In both cases there was a consistent rationale for the changes: the lists were modified to provide a list of impairments specific to the AOC. Some impairments were either not applicable in these areas or any potential uses following delisting were believed to be undesirable. The modified lists also seem to recognize that some of the 14 impairments are not mutually exclusive. For example, loss of fish and wildlife habitat is commonly a cause of degraded fish populations rather than a stand-alone indicator of some particular aspect of ecosystem health.

6.3 ALTERATIONS TO THE LIST OF IMPAIRED BENEFICIAL USES

The applicability of the thirteen impairments described by this study in the Lake Ontario context is shown in Table 6.1; they are not in any particular order. The issues outlined below also seem to have relevance to the Lake Ontario LAMP process based on our assessment of the available information associated with the thirteen impairments and the comments of experts whom we consulted (see Chapter 3). These suggestions are not ranked in any order of priority either.

Table 6.1. Relevance of beneficial use impairments in Lake Ontario.[a]

Listed Impairments Relevant to Lake Ontario
Restrictions on fish and wildlife consumption
Degradation of fish and wildlife populations
Fish tumors or other deformities
Bird or animal deformities or reproduction problems
Degradation of benthos
Restrictions on dredging activities
Eutrophication or undesirable algae
Restrictions on drinking water consumption, or taste and odor
Beach closings
Degradation of aesthetics
Degradation of phytoplankton and zooplankton populations
Listed Impairments Not Relevant to Lake Ontario
Tainting of fish and wildlife flavor
Added costs to agriculture and industry
Suggested Additional Impairments Relevant to Lake Ontario
Undesirable exotic species
Degradation of human health
Sediment and water column contamination
Changes in biomarkers as indicators of chemical stress

[a] Loss of fish and wildlife habitat was not considered in this document.

1. Presence of undesirable exotic species and subsequent control methods (see Chapter 3 and Section 5.12): the principal concern is the relatively recent invasion of zebra mussels and attempts to control them. Chlorination of intakes is being advocated as a treatment method, but inevitably some chlorine escapes

from the pipe, creating a problem in the vicinity of the intake. The possibility of toxic organochlorines forming as a result is unacceptable from an ecological health perspective. Exotic species also have the potential to alter food webs and the ecology of the Great Lakes, usually to the detriment of native, valued species.

2. Degradation of human health including reproductive problems, congenital deformities, disease incidence, and psycho-social aspects of living in a contaminated habitat should be given formal and explicit recognition.

3. Contaminant loadings from point sources (e.g., industry and municipalities) and non-point sources (e.g., agriculture and atmosphere) and their impacts on sediment and water quality should be considered. There are extensive databases which document that a variety of contaminants in sediments and the water column exceed background levels but do not exceed legal standards. Most of these data relate to the offshore waters and do not fit easily within the current list of 14 impaired uses.

4. Changes in biomarkers or biochemical indicators of chemical stress: the absence of overt symptoms of disease does not mean that an impairment has not occurred. Body burdens of contaminant residues sufficient to cause biochemical alterations and/or gentoxic effects presumably reflect some, as yet unknown, metabolic impact. Such indicators of exposure are beginning to be used in fish (e.g., Luxon et al., 1987) and wildlife (see Section 5.5), but standard methodologies and controls are required. There is considerable variation among individuals and between groups of organisms, thus the techniques need to be validated in each

organism taking cognizance of polymorphic responses (confounding factor) versus responses to differences in dose (the factor of interest).

Several impaired uses do not seem to be applicable to the Lake Ontario context (Table 6.1). Tainting of fish and wildlife flavor has not been documented recently in Lake Ontario. In part this may be related to the absence of research, but at least in the AOCs there are no documented reports. There may be added costs to agriculture and industry as a result of impaired water quality, but we were not able to find empirical evidence to support this contention. There are clearly some industries for which there are added costs (e.g., beer and food processing, film), but these costs may be related to the special requirements of the industry even when water quality meets all relevant jurisdictional standards and guidelines. Lastly, it is difficult to separate the degradation of benthos from dredging activity impacts, thus combining the two may be more efficient than maintaining them as separate impairments.

The list of impairments investigated in this report could be rearranged so that they reflect the ecosystem component impacted by contaminants (Table 6.2). For consistency, the additions suggested in Table 6.1 are also shown along with the 13 impairments described here as routes by which contaminants may impair the ecosystem. The list as shown is not comprehensive in the sense that there are probably other ways in which these components can be impaired by contaminants. Furthermore, sediment and water quality could be considered aspects of the biotic impairments in that they at least partially define habitat. This rearrangement logically reflects findings that similar problems in different organisms are caused by or related to the same classes of chemicals. It also redistributes the ecological and

Table 6.2. Impairments of the Lake Ontario ecosystem. The impaired beneficial uses in Annex 2 of the Great Lakes Water Quality Agreement are routes by which contaminants may adversely affect ecosystem components. The italicized pathways are suggestions from Table 6.1.

Impairment of:

Aquatic Biota

- degradation of fish, phytoplankton and zooplankton populations
- fish tumors or other deformities
- degradation of benthos
- loss of habitat
- *presence of exotic species*
- *changes in indicators of chemical stress (biomarkers)*

Wildlife Dependent on Aquatic Biota

- degradation of bird and wildlife populations
- bird or animal deformities or reproductive problems
- loss of habitat
- *changes in indicators of chemical stress (biomarkers)*

Human Health

- restrictions on human consumption of fish and wildlife
- restrictions on drinking water consumption, or taste and odor
- beach closings
- degradation of aesthetics
- *congenital abnormalities, reproductive or developmental effects*
- *psycho-social impacts*

Water Quality

- eutrophication or undesirable algae
- beach closings
- *elevated water-column contaminant concentrations*

Sediment Quality

- degradation of benthos
- restrictions on dredging activities
- *eleveated sediment contaminant concentrations*

jurisdictional impairments into coherent categories. The jurisdictional impairments represent potential routes of human exposure and are so grouped under human health. This rearrangement emphasizes the need to consider the biological component of the Lake Ontario system rather than just the water or sediments when load reductions are contemplated and implemented. The causal pollutants that we listed were drawn from many impaired beneficial uses not just water or sediment quality.

6.4 NEARSHORE VERSUS OFFSHORE IMPAIRMENTS

Most of the impairments relate to nearshore rather than offshore waters and therefore their applicability has to be carefully interpreted in the open waters. The open waters of Lake Ontario are partially separated physically and chemically from nearshore waters during the period when the thermal bar exists (Stevens, 1988), but they have a large mass and fewer direct sources of pollution. For some AOCs the open-waters could be considered a continuum in that pollutants are transported from the AOCs into the less contaminanted open waters by biotic and abiotic mechanisms. We need not expect the same number of impairments to apply in the open waters as in nearshore waters.

The applicability of beach closures, drinking water restrictions, and aesthetic impairments to the offshore waters is not immediately obvious. Beach closures usually relate to swimming, but swimming activities do not require a beach. Thus this impairment could be applied to the open lake if bacteria levels exceed the relevant jurisdictional guidelines (J. Hartig, International Joint Commission, Windsor, Ontario, personal communication), restricting swimming in offshore waters. Similarly, drinking water restrictions could apply

to ship captains wishing to use these waters as a source of drinking water while in transit, if the relevant standards were exceeded (M. Neilson, Environment Canada, Water Quality Branch, Burlington, Ontario, personal communication). Furthermore, oil slicks, algal mats and debris etc. on the open waters may be considered an impairment by recreational boaters. In contrast, impairments such as degradation of phytoplankton, zooplankton, fish, and bird populations, and eutrophication have direct relevance, but in some cases (e.g., phytoplankton and zooplankton populations) the information available is not sufficient to make a confident assessment.

A second parallel approach to assessing the offshore waters with a suite of measures relating to a variety of phenomena is the use of integrative indicators (see Chapter 3). Currently, lake trout and *Pontoporeia hoyi* (an amphipod) have been developed as integrative indicators of oligotrophic conditions in offshore waters, as in the open waters of Lake Ontario (Ryder and Edwards, 1985). Other species such as walleye or *Hexagenia limbata* (a burrowing mayfly) have been proposed for mesotrophic waters such as the Bay of Quinte (Edwards and Ryder, 1990). Integrative indicators in benthos, fish, and wildlife are being developed because they permit the integration of physical and chemical information into a measurable biological effect (Ecosystem Objectives Working Group, 1990a). If these indicators are formally adopted, then they will provide biologically relevant measures of progress in meeting proposed ecological and human health objectives in Lake Ontario.

Our findings and the work of the Ecosystem Objectives Working Group (1990a) emphasize that the biological compartment must be considered when contaminant loadings are determined. Bulk characterization of sediment and water contaminants has little relevance

without reference to potential or known biotic effects. Because aquatic biota and wildlife bioaccumulate and biomagnify contaminants many times greater than in the sediments or water, the relatively small concentrations in these abiotic compartments may result in adverse effects that are difficult to detect, but signal the need for extensive remediation. Such remediation may require changes to the trophic structure of the ecosystem to reduce the biomagnification potential through organisms, e.g., shortening of the food chain to top predators like lake trout (Rasmussen et al., 1990), as well as abatement actions directed at the sources. The pollutants that we designate with confidence as lake-wide causal pollutants are those drawn from many impaired beneficial uses, not just water or sediment quality.

6.5 DATA QUALITY AND ACCESS

Data and information from four disparate scientific fields - analytical measurements, ecology, epidemiology, and toxicology - were collected to describe and catalogue impaired beneficial uses in Lake Ontario (Chapter 5). A further goal of our study was to identify causal pollutants whose elimination or reduced loading will restore beneficial uses (Section 1.2). None of the evidence from the four domains we identified is sufficient on its own to accomplish this goal. However, the combination of analytic, toxicologic, ecologic, and, if available, epidemiologic evidence can be use to reach a weight-of-evidence decision concerning causality. This approach requires knowledge of the reliability and validity of the evidence so that sound judgement as to its use will result. For these reasons, we developed and used critical appraisal guidelines to assess all of the studies cited in this report, with some exceptions as noted. The critical appraisal guidelines were modified to fit each of the

major domains, but all were based on the five generic criteria (see Chapter 4).

For a number of reasons we did not attempt to rate the papers on some kind of global numerical scale or aggregate assessment (e.g., good or bad studies) as suggested at the workshop held 11 June 1991 (see Chapter 3). We assessed each study in a "four-dimensional space", each dimension corresponding to one of the major domains of information mentioned above and only with respect to the information or data relevant to an impaired beneficial use in Lake Ontario. None of the studies that we reviewed was written from this perspective. Our assessments of a study might well be different if we had started from the study's own perspective. Hence imposing a rating system on the critical appraisal would be a highly biased and arbitrary judgement of a paper and out of the context of our study. Furthermore, the critical appraisal guidelines were designed to highlight the strengths and weaknesses of individual studies. Crediting or discrediting a particular study is incompatible with this objective and as such was rejected from the beginning of the process. We have tried to convey our findings concerning the strengths and weaknesses of studies as relevant to our work in a generic sense and in a constructive manner below, with the expectation that they may help to improve environmental science as practiced in the Great Lakes.

An essential prerequisite of our study was the use of long-term databases, covering physical, chemical, and biotic aspects of the Lake Ontario ecosystem. Databases made available to us document lake trout abundance in eastern Lake Ontario (Ontario Ministry of Natural Resources, 1991; New York Department of Environmental Conservation, 1991), contaminant residues in spottail shiners (Skinner and Jackling, 1989; Suns et al., 1991), contaminant residues in sport fish (Sloan et al., 1987; OME and OMNR, 1991) and lake trout

(Borgmann and Whittle, 1991; Whittle and Keir, 1991), contaminant residues in herring gull eggs (e.g., Bishop et al., 1991), contaminants in and quantities of dredged sediments by project and location (Dredging Subcommittee, 1982; Sediment Work Group, 1990, 1991), and changes in water quality parameters, including trophic status indicators, of Lake Ontario waters (e.g., Stevens, 1988). This incomplete listing represents some of the more useful databases in the context of our study. For the most part these databases have been compiled using internally consistent sampling protocols and well documented, appropriate quality control procedures. In recent years more attention has been devoted to describing more precisely the sampling methodologies and analytical procedures employed in the compilation of these databases. More researchers have become aware of the need for QA/QC procedures, particularly with respect to contaminant analysis, and have incorporated them into their research. We view these as positive developments and encourage their widespread adoption.

In spite of these strengths there are weaknesses in the data as a whole which complicated our task of identifying causal pollutants in Lake Ontario. There is a dismaying lack of standardization in terms of sampling, sample storage, analytical methodologies, and statistical treatment of the data. To some extent this is the inevitable consequence of a diverse array of research and research objectives, but it prevents an accurate picture of current conditions in Lake Ontario. For example, the histopathologic criteria used to diagnose fish tumors have been standardized in Canada but not between Canada and the United States (Section 5.4). The sediment guidelines used to evaluate dredged sediments for open-water disposal differ in terms of the criteria and their application between Canada and the United States (Section 5.7), although standardization is being fostered by the Sediment Work Group

(1990, 1991) of the IJC. A more visible example relates to the differences between the Ontario and New York sport fish contaminant programs and the resulting advice to consumers (Section 5.1). The need for standardization to reduce variability and promote harmonization for interpretation of results is apparent.

Many of the datasets that we used are discrete, unpublished, and very personalized. There is no central registry of information for the Great Lakes indicating the type of data collected and an appropriate contact person. In part this is the reason that our initial information and data gathering stage relied on interviews with individuals who have expertise on Great Lakes issues (see Appendix A for a complete list of contacts and Chapter 3). We interviewed over 100 experts to identify the major databases and sources of information pertinent to Lake Ontario. Near the end of this phase the process became circular in that we already were aware of the suggested contacts and data sources from previous interviews. We therefore had confidence that we had identified most of the major databases and relevant information, which we subsequently checked with some of the same experts at a workshop held on 11 June 1991 (see Chapter 3 for a workshop summary).

Although a central registry of information would alleviate some of the access problems, there remains a proprietary problem, especially but not exclusively with respect to data from industries. Those who collect data tend to view them as "their own", to use or misuse as they choose. This attitude is fostered by an antagonistic, quasi-judicial, regulatory system and persists in certain limited areas. But it is being rapidly superseded by a more open, pluralistic, less antagonistic system of environmental management. This new system is consistent with an ecosystem approach and there are optimisitic expectations that the RAP

process in the Great Lakes is the forerunner of this new system.

6.6 CAUSE-EFFECT ARGUMENTS AND WEIGHT-OF-EVIDENCE DECISIONS

Great Lakes people have staggered through a series of environmental "crises" repeatedly since the turn of the century. By the late 1950s sea lamprey predation combined with overfishing had led to the collapse of commercial fisheries for valued salmonines such as lake trout, in all of the lakes. In the 1960s eutrophication dominated the attention of regulatory officials and the research community. By the late 1960s the southerly half of the Great Lakes was severely enriched and contaminated, the latter through the use of industrial, agricultural, and household chemicals and through the indiscriminate loading of wastes from their production into the environment. The earlier problems were successfully attacked using conventional reductionistic science, i.e., chemical-analytical and toxicological research in the laboratory together with engineering technology, e.g., devising better sewage treatment and phosphate-free detergents. This was not so with contamination, particularly organochlorine contamination. Earlier problems were essentially linear in the sense that there were only a few causes and a few effects, with nearly direct links between them. The laboratory/engineering approach is well suited to this type of problem and it was appropriate for these tasks until the 1960s. In the context of contamination, conventional science is still necessary because it defines the "potential effects" in the biota, but it is not sufficient because it cannot link actual damage to cause. A problem such as organochlorine contamination has complex linkages between multiple cause and multiple effects at different levels of biological organization and thus cannot be specified simply. Simple cause-and-effect linkages are

therefore not likely.

In recent years case studies have been published in which damage to aquatic biota or wildlife was inferred to be caused by chemicals (see Colborn et al., 1990; Gilbertson and Schneider, 1991). These studies are based on modified epidemiological criteria similar to those used in this study (see Chapter 4), which were used to infer causality based on the weight-of-evidence with information from a diversity of sources. The collective experience of these studies combined with the findings of our study may be relevant to the Lake Ontario LAMP as sketched below.

1. The effects of contaminants are similar across a variety of organisms and biological levels of organization and some evidence indicates that these effects extend generally to humans.

2. Organochlorine contaminants most commonly are associated with reproductive and developmental dysfunctions and therefore must be considered teratogenic compounds. Many of the regulatory programs in the Great Lakes employ carcinogenic endpoints in the setting of criteria, particularly for the protection of human health (e.g., see Sections 5.1 and 5.9). The case studies of damage to fish and wildlife in the Great Lakes demonstrate that teratogenic outcomes constitute important biological endpoints.

3. A single contaminant approach generally does not work with, for example, organochlorine contaminants in an ecological context. This approach would ignore synergistic or possible antagonistic interactions among the suite of chemicals in the environment. Because of the difficulty in singling out specific causal pollutants, it might be more appropriate to focus on families or classes of similar chemicals based on their

toxicological profiles, e.g., PCBs, dioxins, or PAHs, as a starting point. It is not always possible to isolate a direct linkage between cause and effect; in these instances cause-effect associations (weight-of-evidence arguments) using all of the available information from toxicology, epidemiology, physical/chemical analysis, and ecology must be sufficient. This should not be used to question the need for abatement activities directed at PCBs, dioxins/furans, etc. We now know enough about the harmful aspects of some of these substances to get on with the job of reducing their loadings into Lake Ontario.

It is our view that a singular emphasis on specific congeners (e.g., coplanar PCBs) seems somewhat risky, although currently vogue. This toxicologic paradigm assumes a clear understanding of the determinants or pathogenic steps involved in such multifactorial (often polygenic) diseases such as cancer or teratogenic effects. It may well raise false hopes of establishing unequivocal cause-and-effect linkages, and in this manner delay preventive and corrective actions. A weight-of-evidence approach to formulating a level of concern and action based on empirical (field) evidence of damage, group toxicologic profiles, and actual input/loading data seems more prudent.

4. The contaminants themselves do not follow geographical or political boundaries. For example, mirex was apparently only produced and processed in the Lake Ontario watershed (Holdrinet et al., 1978), yet it is detectable in common terns and herring gulls from colonies in the upper Great Lakes (e.g., Weseloh et al., 1989, 1990) that over-winter in Lake Ontario habitats. It is not now possible to build a convincing association or cause-effect case based on evidence from a single lake. The Great Lakes basin must be treated as a whole, using evidence from the other Lakes and from other sources and locales to construct a

compelling case.

5. Concentrations of contaminants in the water or sediment have little meaning by themselves without reference to biotic effects. Some contaminants have been inferred from laboratory tests to be hazardous at any level and for these any amount is unacceptable (e.g., the GLWQA specifies that mirex should be "virtually absent" in aquatic biota). For some pollutants (e.g., phosphates) a maximal level has been specified as acceptable. In both cases the levels used are based on biological effects. Loading reductions must therefore consider the consequences in terms of the biota, water, and sediments of the Lake Ontario ecosystem.

6. Until recently the burden of proof concerning possible adverse effects of artificial materials did not lie with the creator and distributor of these materials in aquatic systems. The onus is now shifting politically; we may be in a transitional period. The burden of proof that a chemical has no significant short- or long-term adverse effects may have to be met by a manufacturer prior to its production or use within the environment. This call is consistent with work on "sunset" provisions for chemicals (National Wildlife Federation/Canadian Institute for Environmental Law and Policy, 1991) and with the requirement of the amended GLWQA to virtually eliminate the input of persistent toxic substances into the Great Lakes basin (Virtual Elimination Task Force, 1991).

6.7 CONCLUDING COMMENTS

The application of critical appraisal guidelines to the available data on Lake Ontario has resulted in the identification of locations where impaired beneficial uses occur and a list of causal pollutants/agents. Both of these lists are heavily influenced by the intensity of

research work which is dictated in part by the management framework, such as RAPs, and in part by individual and agency interests and perspectives on the importance of, for example, fish contamination versus benthic contamination. The other major factor influencing our findings is the relative ease in determining impairments based on exceedances of jurisdictionally-specified guidelines or standards compared to the scientific judgement and multiple information sources required to establish impairment of ecologically-based beneficial uses.

The following two paragraphs are excerpted from Bulkley et al. (1989) and illustrate the transitional period that we may now be in.

Gunnerson (1988) has provided an interesting distinction between two kinds of severe environmental difficulties. Where the issue is well-understood scientifically, where sufficient technological capabilities exist and where the institutional mechanisms for program delivery are in place, there the problem may be viewed as big and bad - like Godzilla of the 1960s movie - but it can be solved in the conventional way through commitment of sufficient resources. The more quickly resources are committed and mobilized, the more rapid the progress toward the solution. This Godzilla model fits approximately the attempt under the 1972 GLWQA to reverse the eutrophication process that was caused predominately by cultural loadings of phosphates into the waters.

In contrast, and with the benefit of hindsight, we note that the main issue of the 1978 GLWQA - hazardous chemical contaminants - was poorly understood scientifically, could not be resolved fully by current technology and did not have available the appropriate institutional mechanisms for its resolution. Here political will as reflected only in a

willingness by the federal parties to commit sufficient conventional resources would not likely succeed: a more comprehensive mobilization of political will would have been necessary. Gunnerson (1988) suggested that a problem such as this was analogous to the Andromeda Strain, a 1969 novel and later a film.

A growing movement to take lakewide management planning seriously in Lake Ontario and elsewhere may indicate sufficient political commitment to deal effectively with hazardous chemicals, however difficult the task.

CHAPTER 7 RECOMMENDATIONS

1. The use of a critical appraisal approach to review relevant literature is essential to the reliable selection of causal pollutants and hence the development of the LAMP. This approach permits an objective analysis of the information on impairments of beneficial uses in Lake Ontario. The information then provides the building blocks leading to the list of causal pollutants. The critical appraisal questions may be adjusted to fit any similar study.

2. Sampling methods, storage requirements, and analysis of contaminants needs further standardization among parties. The differences now prevent definitive comparisons among data collected by different workers or over different time periods. This lack of consistency provides opportunity for postponing action for as long as the different agencies care to quarrel about apparent inconsistencies in findings that are due in part to lack of standardized methods.

3. Emphasis should be placed on developing a few, long-term databases to assess the state-of-the-lake. The current assortment of efforts prevents a fully comprehensive assessment of the occurrence of the impairments. Ideally, the data bases would include the entire Lake Ontario basin with binational participation.

4. Improved data access and communication is required. Most of the data on Lake Ontario are held within agencies in raw form and are unpublished. While the agencies may make this information available upon request, it is difficult to get an overview of the impairments based on these raw data. The raw data cannot be presented without specific details concerning the rationale, sampling and analytical protocols, and QA/QC procedures. Futhemore, these data may not have been be collected with the goal of evaluating impairments of beneficial uses. Even when published, the documents may not be widely distributed, so collecting information on impairments becomes a system of referrals to scientists working in individual areas. While this does provide valuable insights, it is a time-consuming and inefficient method of using the data collected.

5. Integration of scientific efforts, to produce comprehensive understanding needs increased emphasis. While there have been notable efforts towards integrative, cooperative research such as the International Field Year on Lake Ontario and the Niagara River Toxics Committee report, much more synergy is needed among research areas. For example, benthic macroinvertebrate community structure is not often correlated with chemical contaminant concentrations as would be necessary to assess the impairment based on degradation of benthos. Increased synergy is also required at the binational level to limit overlapping efforts and channel energy into toxic load reductions.

6. Guidelines for additional contaminants need to be developed based on appropriate research, some of which might be new. This would potentially increase the number of causal pollutants. The listing/delisting guidelines for impairments of beneficial uses can be divided into two types: ecologically-based impairments such as bird and animal deformities and jurisdictionally-based impairments using exceedences of standards, guidelines and criteria such as dredging restrictions. Due to the nature of proof, it is easier to identify jurisdictionally-based impairments than ecologically based impairments. For example, the setting of new guidelines for organics in dredging would increase the likelihood that these would then be found to be impaired with respect to dredging and so be perceived as potential causal pollutants.

7. Several additions should be made to the current list of impairments of beneficial uses. The presence of exotic species such as zebra mussels is not specifically included in any of the existing impairments. Exotics have the potential to significantly alter the existing ecological links and so have a profound effect on nutrient and contaminant cycling and on phytoplankton, benthic, zooplankton, fish, and human populations. Their role should be explicitly recognized.

8. Degradation of human health should be explicitly recognized as a new impairment. Human health effects are indirectly recognised through impairments on fish consumption, drinking water, and beach closures. The evidence from other organisms and biological levels of organization suggests that the effects of contaminants extend generally to humans.

9. Biochemical indicators of stress in wildlife and fish could be added to either of two existing impairments: degradation of fish and wildlife populations; or bird and animal deformities and reproductive problems. This broadening of the definition of an existing impairment would recognize the increasing information provided by biochemical markers of continuing alterations in Great Lakes wildlife.

10. The existing list of impairments could be reorganized in a more systematic form to reflect the ecosystem components impacted by contaminants.

11. Most of the impairments that we documented occurred in the RAP areas, reflecting the intensive government and public effort in these areas. The impairments in the open lake appear to be less frequent, but this may be an artifact of less intensive scientific effort, rather than an accurate reflection of the true situation.

12. The development of Lakewide Management Plans should treat the Lake as an ecosystem. This would mean including the impairments from the nearshore zone and Areas of Concern. Inputs from the nearshore zone influence the impairments in the open lake waters, and the quality of the water in the open lake also influences nearshore conditions during turnover periods in the Lake. Therefore, the LAMP should include both the nearshore and offshore zones in developing loading reductions of critical pollutants.

13. The applicability of specific traditional nearshore impairments should be clarified. The International Joint Commission has stated that impairments of beach closures or drinking water restrictions could occur in the open lake if contaminant levels exceed relevant guidelines. This position needs to be confirmed by the Parties to the Agreement.

14. The development of integrative ecological indicators for Lake Ontario should be expedited. Some species integrate much of the physical and chemical information relevant to list of impaired uses into a measurable biological effect, and so provide ecosystemic information directly relevant to the purpose of the Agreement.

15. If the United States Environmental Protection Agency develops an inventory of impairments focusing on the southern shore of Lake Ontario, we suggest that a methodology generally similar to the one used here be followed. This would ensure consistency, minimize confusion, and increase the utility of the lists of causal pollutants.

16. The 22 causal pollutants identified through this study should be considered as candidates for critical pollutants for Lake Ontario. Some pollutants such as PCBs, PAHs, and TCDD are associated with more than one impairment, which increases confidence in their selection as a critical pollutant. Four of these causal pollutants are surrogate measures, and may be best used as indicators of chemical or nutrient contamination.

CHAPTER 8. LITERATURE CITED

Advisory Council on Occupational Health and Safety. 1983. Advisory memorandum 82-V to the Minister of Labour, February 14, 1983, Principles and Procedures for the Interpreation of Epidemiological Studies. Fifth Annual Report, Ontario Ministry of Labour, Toronto, Ontario, p. 135-145.

American Public Health Association, American Water Works Association, and the Water Pollution Control Federation. 1985. Standard methods for the examination of water and wastewater. Sixteenth edition. American Public Health Association, Washington, D.C., xlix + 1268 pp.

Amoore, J.E., and Theimer, E.T., [ed.]. 1982. Odour Theory and Odour Classification. Fragrance Chemistry. Academic Press, New York.

Armstrong, R.W., and Sloan, R.J. 1980. Trends in Levels of Several Known Chemical Contaminants in Fish from New York State Waters. New York State Department of Environmental Conservation, Division of Fish and Wildlife, Bureau of Environmental Protection, Albany, New York, Technical Report 80-2, 77 pp.

Aubert, E.J., and Richards, T.L. [ed.]. 1981. IFYGL - The International Field Year for the Great Lakes. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Great Lakes Environmental Research Laboratory, Ann Arbor, Michigan, 410 pp.

Aulerich, R.J., and Ringer, R.K. 1977. Current status of PCB toxicity and mink, and effect on their reproduction. Archives of Environmental Contamination and Toxicology 6:279-292.

Barlow, S.M., and Sullivan, F.M. 1982. Reproductive Hazards of Industrial Chemicals, Academic Press, London, pp. 9-27.

Bartels, J.H.M., Burlingame, G.A., and Suffet, I.H. 1986. Flavor profile analysis: taste and odour control of the future. Journal of the American Waterworks Association, March issue:50-55.

Baumann, P.C. 1984. Cancer in wild freshwater fish populations with emphasis on the Great Lakes. Journal of Great Lakes Research 10:251-253.

Baumann, P.C., and Whittle, D.M. 1988. The status of selected organics in the Laurentian Great Lakes: an overview of DDT, PCBs, dioxins, furans, and aromatic hydrocarbons. Aquatic Toxicology 11:241-257.

Baumann, P.C., Harshbarger, J.C., and Hartman, K.J. 1990. Relationship between liver tumors and age in brown bullhead populations from two Lake Erie tributaries. Science of the Total Environment 94:71-87.

Baumann, P.C., Smith, W.D., and Parland, W.K. 1987. Tumor frequencies and contaminant concentrations in brown bullheads from an industrialized river and a recreational lake. Transactions of the American Fisheries Society 116:79-86.

Bay of Quinte RAP Team. 1990. Bay of Quinte Remedial Action Plan. Stage I. Environmental Setting and Problem Definition. Ontario Ministry of the Environment, Eastern Region, Kingston, Ontario, 236 pp + appendices.

Beak. 1989. St. Catharines Area Pollution Control Plan, River Modelling, Beak Consultants, St. Catharines, Ontario.

Beeton, A.M. 1965. Eutrophication of the St.Lawrence Great Lakes. Limnology and Oceanography 10:240-254.

Bellis, V.J., and McLarty, D.A. 1967. The ecology of *Cladophora glomerata* (L.) Kutz. in Southern Ontario. Journal of Phycology 3:57-63.

Bertram, P., [ed.]. 1985. Limnology and Phytoplankton Structure in Nearshore Areas of Lake Ontario, 1981. United States Enivronmental Protection Agency, Great Lakes National Program Office, Chicago, Illinois, EPA-905/3-85-003, 172 pp.

Binational Objectives Development Committee. 1989. Great Lakes Water Quality Agreement, Annex 1, Three Lists of Substances, Draft 1989 Issue. Binational Objectives Development Committee, International Joint Commission, Windsor, Ontario.

Bishop, C., Brooks, R.J., Carey, J.H., Ng, P., Norstrom, R.J., and Lean, D.R.S. 1991a. The case for a cause-effect linkage between environmental contamination and development in eggs of the common snapping turtle (*Chelydra s.serpentina*) from Ontario, Canada. Journal of Toxicology and Environmental Health 33:521-547.

Bishop, C.A., Weseloh, D.V., Burgess, N.M., Struger, J., and Norstrom, R.J. 1991b. An atlas of contaminants in eggs of colonial fish-eating birds of the Great Lakes, 1970-1988. Environment Canada, Canadian Wildlife Service, Burlington, Ontario, (in press).

Black, J.J. 1983. Field and laboratory studies of environmental carcinogenesis in Niagara River fish. Journal of Great Lakes Research 9:326-334.

Black, J.J. 1984. Environmental implications of neoplasia in Great Lakes fish. Marine Environmental Research 14:529-534.

Black, J.J. 1988. Fish tumors as known field effects of contaminants. *In* Schmidtke, N.W., [ed.], Toxic Contamination in Large Lakes. Volume 1. Chronic Effects of Toxic Contaminants in Large Lakes, Lewis Publishers, Chelsea, Michigan, pp. 55-81.

Blokpoel, H., and Tessier, G.D. 1991. Distribution and Abundance of Colonial Waterbirds Nesting in the Canadian Portions of the Lower Great Lakes System in 1990. Canadian Wildlife Service, Ontario Region, Technical Report Series No. 117, 16 pp.

Borgmann, U., and Whittle, D.M. 1991. Contaminant concentration trends in Lake Ontario lake trout (*Salvelinus namaycush*): 1977 to 1988. Journal of Great Lakes Research 17:368-381.

Botts, L., and Krushelnicki, B. 1987. The Great Lakes. An Environmental Atlas and Resource Book. Published jointly by Environment Canada, U.S. Environmental Protection Agency, Brock University, and Northwestern University, 44 pp.

Boyce, F.M., Donelan, M.A., Hamblin, P.F., Murthy, C.R., and Simons, T.J. 1989. Thermal structure and circulation in the Great Lakes. Atmosphere-Ocean 27:607-642.

Braune, B.M., and Norstrom, R.J. 1989. Dynamics of organochlorine compounds in herring gulls: III. Tissue distribution and bioaccumulation in Lake Ontario gulls. Environmental Toxicology and Chemistry 8:957-968.

Brown, T.L., Knuth, B.A., and Menz, F.C. 1991. Lake Ontario's sport fisheries: socioeconomic research progress and needs. Canadian Journal of Fisheries and Aquatic Sciences 48:1595-1601.

Brownlee, B.G., Painter, D.S., and Boone R.J. 1984. Identification of taste and odour compounds from Western Lake Ontario. Water Pollution Research Journal of Canada 19:111-118.

Bulkley, J.W., Donahue, M.J., and Regier, H.A. 1989. The Great Lakes Water Quality Agreements: How to assess progress toward a goal of ecosystem integrity. *In* Gunnerson, C.G., [ed.], Post-audits of environmental programs and projects. American Society of Civil Engineers, New York, pp. 27-42.

Cairns, V.W., Hodson, P.V., and Nriagu, J.O., [ed.]. 1984. Contaminant effects on fisheries. Advances in Environmental Science and Technology, Volume 16. John Wiley and Sons, New York, xv + 333 pp.

CanTox, SENES, and Ontario Ministry of the Environment. 1986. Priority List Working Group. Part I Report: Vector Scoring System for the Prioritization of Chemical Contaminants. Draft Report for the Ontario Ministry of the Environment, Toronto, Ontario.

Canviro. 1989a. Final Report: St. Catharines Area Pollution Control Plan - Urban and Industrial Discharges Study, Canviro Consultants Report Prepared for the Ontario Ministry of the Environment, Toronto, Ontario.

Canviro. 1989b. Thirty-seven Municipal Water Pollution Control Plants, Pilot Monitoring Study, Volume I, Interim Report, Canviro Consultants Report Prepared for the Ontario Ministry of the Environment, Toronto, Ontario, 99 pp.

Chong, B. 1984. The Status of Public Bathing Beaches on the Great Lakes Shoreline. Draft for Ontario Ministry of the Environment. Toronto, Ontario, 95 pp.

Christie, W.J. 1972. Lake Ontario: effects of exploitation, introductions, and eutrophication on the salmonid community. Journal of the Fisheries Research Board of Canada 29:913-929.

Christie, W.J. 1973. A Review of the Changes in the Fish Species Composition of Lake Ontario. Great Lakes Fishery Commission Technical Report No. 23, 65 pp.

Christie, W.J. 1974. Changes in the fish species composition of the Great Lakes. Journal of the Fisheries Research Board of Canada 31:827-854.

Christie, W.J., Scott, K.A., Sly, P.G., and Strus, R.H. 1987. Recent changes in the aquatic food web of eastern Lake Ontario. Canadian Journal of Fisheries and Aquatic Sciences 44(Supplement 2):37-52.

Clement International Corporation. 1990. Toxicological Profile for Polycyclic Aromatic Hydrocarbons. U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry, Washington, D.C., TP-90-20, 231 pp.

Colborn, T. 1989a. Bald eagles in the Great Lakes Basin. *In* Gilbertson, M., [ed.], Proceedings of the Workshop on Cause-effect Linkages. International Joint Commission, Council of Great Lakes Research Managers, Windsor, Ontario, pp. 10-11.

Colborn, T. 1989b. Innovative approaches for evaluating human health in the Great Lakes basin using wildlife toxicology and ecology. Manuscript prepared for the International Joint Commission, Windsor, Ontario, 56 pp.

Colborn, T.E., Davidson, A., Green, S.N., Hodge, R.A., Jackson, C.I., and Liroff, R.A. 1990. Great Lakes Great Legacy? The Conservation Foundation, Washington, D.C., and the Institute for Research on Public Policy, Ottawa, Ontario, xliii + 301 pp.

Colborn, T. 1991. Epidemiology of Great Lakes bald eagles. Journal of Toxicology and Environmental Health 33:395-453.

Cox, C., and Johnson, A.F. 1990. A Study of the Consumption Patterns of Great Lakes Salmon and Trout Anglers. Ontario Ministry of the Environment, Water Resources Branch, Sport Fish Contaminant Monitoring Program, Toronto, Ontario, v + 19 pp.

Cox, C., and Ralston, J. 1990. A Reference Manual of Chemical Contaminants in Ontario Sport Fish. Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario, v + 32 pp.

Daly, H.B., Hertzler, D.R., and Sargent, D.M. 1989. Ingestion of environmentally contaminated Lake Ontario salmon by laboratory rats increases avoidance of unpredictable aversive nonreward and mild electric shock. Behavioral Neuroscience 103:1356-1365.

De Vault, D., Dunn, W., Berqvist, P.-E., Wiberg, K., and Rappe, C. 1989. Polychlorinated dibenzofurans and polychlorinated dibenzo-*p*-dioxins in Great Lakes fish: a baseline and interlake comparison. Environmental Toxicology and Chemistry 8:1013-1022.

Department of Health, Education and Welfare. 1964. Smoking and Health. United States Department of Health, Education and Welfare, Washington, D.C., Public Health Service Publication No. 1103.

Dickman, M.D., and Steele, P.O. 1986. Gonadal neoplasms in wild carp-goldfish hybrids from the Welland River near Niagara Falls, Canada. Hydrobiologia 134:257-263.

Dobson, H.F.H. 1984. Lake Ontario Water Chemistry Atlas. Environment Canada, Inland Waters Directorate, Burlington, Scientific Series No. 139, xiii + 59 pp.

Down, N.E., Peter, R.E., and Leatherland, J.F. 1988. Gonadotropin content of the pituitary gland of gonadal tumor-bearing common carp x goldfish hybrids from the Great Lakes, as assessed by bioassay and radioimmunoassay. General and Comparative Endocrinology 69:288-300.

Dredging Subcommittee. 1982. Guidelines and Register for Evaluation of Great Lakes Dredging Projects. Report to the Water Quality Programs Committee of the Great Lakes Water Quality Board, International Joint Commission, Windsor, Ontario, ix + 365 pp.

Duffus, J.H., and Richardson, M.L. 1990. Glossary of Terms for Chemical Toxicology and Ecotoxicology. International Union of Pure and Applied Chemistry, Clinical Chemistry Division, Commission on Toxicology Working Party, London, U.K., 172 pp.

Durham, R.W., and Joshi, S.R. 1980. Investigation of Lake Ontario water quality near Port Granby Radioactive Waste Management Site. Water, Air, and Soil Pollution 13:17-26.

Durham, R.W., and Oliver, B.G. 1983. History of Lake Ontario contamination from the Niagara River by sediment radiodating and chlorinated hydrocarbon analysis. Journal of Great Lakes Research 9:160-168.

Ecological Objectives Committee. 1989. The Bald Eagle. Report to the Science Advisory Board, International Joint Commission, Windsor, Ontario, 12 pp.

Ecosystem Objectives Work Group. 1990a. Ecosystem Objectives for Lake Ontario. Prepared by the Parties to the Great Lakes Water Quality Agreement, International Joint Commission, Windsor, Ontario, 14 pp.

Ecosystem Objectives Work Group. 1990b. The Development of an Ecosystem Health Indicator for Lake Ontario Based on Benthic Macroinvertebrate Communities. Canada Centre for Inland Waters, Dec. 10/11, 1990.

Edwards, C.J., and Ryder, R.A., [ed.]. 1990. Biological Surrogates of Mesotrophic Ecosystem Health in the Laurentian Great Lakes. Report to the Science Advisory Board, International Joint Commission, Windsor, Ontario, viii + 69 pp.

Ellenton, J.A., and McPherson, M.F. 1983. Mutagenicity studies on herring gulls from different locations on the Great Lakes. I. Sister chromatid exchange rates in herring gull embryos. Journal of Toxicology and Environmental Health 12:317-324.

Ellenton, J., Brownlee, L, and Hollebone, B. 1985. Aryl hydrocarbon hydroxylase levels in herring gull embryos from different locations in the Great Lakes. Environmental Toxicology and Chemistry 4:615-622.

Envirosearch. 1981. Bacteriological Water Quality Along the Toronto Waterfront. Draft report prepared for the Ontario Ministry of the Environment, Toronto, Ontario, 127 pp.

Fitzsimons, J.D. 1987. Hepatocellular adenoma in the slimy sculpin *Cottus cognatus* Richardson. Journal of Fish Diseases 11:89-91.

Foley, R.E., and Batcheller, G.R. 1988. Organochlorine contaminants in common goldeneye wintering on the Niagara River. Journal of Wildlife Management 52:441-445.

Foley, R.E., Jackling, S.J., Sloan, R.J., and Brown, M.K. 1988. Organochlorine and mercury residues in wild mink and otter: comparison with fish. Environmental Toxicology and Chemistry 7:363-374.

Foran, J.A., and VanderPloeg, D. 1989. Consumption advisories for sport fish in the Great Lakes basin: jurisdictional inconsistencies. Journal of Great Lakes Research 15:476-485.

Fox, G. 1989. Introductory methodology. *In* Gilbertson, M. [ed.], Proceedings of the Workshop on Cause Effect Linkages, Chicago, Ilinois, March 28-30. Council of Great Lakes Research Managers, International Joint Commission, Windsor, Ontario, p. 6.

Fox, G. 1991. Biomarkers: What are they and what have they told us about the effects of contaminants on the health of Great Lakes Wildlife. Cause-effects Linkages II Symposium. Sept. 27-28, 1992. Grand Traverse Resort, Traverse City, Michigan.

Fox, G., and Dickson, E.J. 1989. An economic analysis of conservation tillage as a sediment control strategy on southwestern Ontario cropland. Water Pollution Research Journal of Canada 24: 265-277.

Fox, G., and Dickson, E.J. 1990. The economics of erosion and sediment control in southwestern Ontario. Canadian Journal of Agriculture Economics 38:23-44.

Fox, G.A., Kennedy, S.W., and Norstrom, R.J. 1988. Porphyria in herring gulls: a biochemical response to chemical contamination of Great Lakes food chains. Environmental Toxicology and Chemistry 7:831-839.

Fox, G.A., Weseloh, D.V., Kubiak, T.J., and Erchman, T.C. 1991a. Reproductive outcomes in colonial fish-eating birds: A biomarker for developmental toxicants in Great Lakes food chains. I. Historical and ecotoxicological perspectives. Journal of Great Lakes Research 17:153-157.

Fox, G.A., Collins, B., Hayakawa, E., Weseloh, D.V., Ludwig, S.P., Kubiak, T.J., and Erchman, T.C. 1991b. Reproductive outcomes in colonial fish-eating birds: A biomarker for developmental toxicants in Great Lakes food chains. II. Spatial variation in the occurrence and prevalence of bill defects in young double-crested cormorants in the Great Lakes, 1979-1987. Journal of Great Lakes Research 17:158-167.

FPWGRWQ. 1983. Guidelines for Canadian Recreational Water Quality, -Federal-Provincial Working Group on Recreational Water Quality of the Federal-Provincial Advisory Committee in Environmental and Occupational Health, Ottawa, Ontario.

Frank, R., Thomas, R.L., Holdrinet, M., Kemp, A.L.W., and Braun, H.E. 1979. Organochlorine insecticides and PCB in surficial sediments (1968) and sediment cores from Lake Ontario. Journal of Great Lakes Research 5:18-27.

Frank, R., Thomas, R.L, Holdrinet, M.V.H., and Damiani, V. 1980. PCB residues in bottom sediments collected from the Bay of Quinte, Lake Ontario, 1972-1973. Journal of Great Lakes Research 6:371-376.

Frimeth, J. 1990. Lake Ontario coho. *In* Proceedings of the expert consultation on contaminant caused reproductive problems in salmonids. Report of the Ecological Committee, Science Advisory Board, to the International Joint Commission, Windsor, Ontario.

Gilbertson, M. 1974. Pollutants in breeding herring gulls in the lower Great Lakes. Canadian Field-Naturalist 88:273-280.

Gilbertson, M. 1982. Etiology of Chick Edema Disease in Herring Gulls in the Lower Great Lakes. Canadian Technical Report of Fisheries and Aquatic Sciences 1120, ii + 14 pp.

Gilbertson, M. 1986. Etiology of chick edema disease in herring gulls in the lower Great Lakes. Chemosphere 12:357-360.

Gilbertson, M., [ed.]. 1989a. Proceedings of the Workshop on Cause Effect Linkages, Chicago, Illinois, March 28-30, Council of Great Lakes Research Managers, International Joint Commission, Windsor, Ontario, 37 pp.

Gilbertson, M. 1989b. Bald eagle recovery plan. Manuscript prepared for the Ecosystems Objectives Working Group, Science Advisory Board, International Joint Commission, Windsor, Ontario, 25 pp.

Gilbertson, M., and Fox, G.A. 1977. Pollutant-associated embryonic mortality of Great Lakes herring gulls. Environmental Pollution 12:211-216.

Gilbertson, M., and Hale, R. 1974a. Early embryonic mortality in a herring gull colony in Lake Ontario. Canadian Field-Naturalist 88:354-356.

Gilbertson, M., and Hale, R. 1974b. Characteristics of the breeding failure of a colony of herring gulls in Lake Ontario. Canadian Field-Naturalist 88:356-358.

Gilbertson, M., and Schneider, R.S., [ed.]. 1991. Proceedings of the Workshop on Cause-Effect Linkages. Journal of Toxicology and Environmental Health 33:359-547.

Gilbertson, M., Morris, R.D., and Hunter, R.A. 1976. Abnormal chicks and PCB residue levels in eggs of colonial birds on the Lower Great Lakes (1971-73). Auk 93:434-442.

Gilbertson, M., Kubiak, T., Ludwig, J., and Fox, G. 1991. Great Lakes embryo mortality, edema, and deformities syndrome (GLEMEDS) in colonial fish-eating birds: similarity to chick-edema disease. Journal of Toxicology and Environmental Health (in press).

Gilman, A.P., Fox, G.A., Peakall, D.B., Teeple, S.M., Carroll, T.R., and Haymes, G.T. 1977. Reproductive parameters and egg contaminant levels of Great Lakes herring gulls. Journal of Wildlife Management 41:458-468.

Government of Canada. 1991a. Toxic Chemicals in the Great Lakes and Associated Effects. Volume I - Contaminant Levels and Trends. Environment Canada, Department of Fisheries and Oceans, and Health and Welfare Canada, Ottawa, Ontario, pp. i-vii + 1-488 pp.

Government of Canada. 1991b. Toxic Chemicals in the Great Lakes and Associated effects. Volume II - Effects. Environment Canada, Department of Fisheries and Oceans, and Health and Welfare Canada, Ottawa, Ontario, pp. 489-755.

Gray, I.M. 1987. Differences between nearshore and offshore phytoplankton communities in Lake Ontario. Canadian Journal of Fisheries and Aquatic Sciences 44:2155-2163.

Great Lakes Science Advisory Board. 1989. 1989 Report. Report to the International Joint Commission, Windsor, Ontario, xiv + 92 pp.

Great Lake United. 1988. A Citizens' Agenda for Restoring Lake Ontario. Great Lakes United, Buffalo, New York, 47 pp.

Great Lakes Water Quality Agreement. 1989. Great Lakes Water Quality Agreement (as amended November, 1987). International Joint Commission, Windsor, Ontario, 84 pp.

Great Lakes Water Quality Board. 1985. 1985 Report on Great Lakes Water Quality. Report to the International Joint Commission, Windsor, Ontario, xii + 212 pp.

Great Lakes Water Quality Board. 1989a. 1989 Report on Great Lakes Water Quality. Report to the International Joint Commission, Windsor, Ontario, xxviii + 128 pp.

Great Lakes Water Quality Board. 1989b. 1987 report on Great Lakes water quality: Appendix B - Great Lakes Surveillance, Volume I. Great Lakes Water Quality Board, International Joint Commission, Windsor, Ontario.

Great Lakes Water Quality Board. 1989c. 1987 report on Great Lakes water quality: Appendix B - Great Lakes Surveillance, Volume III. Great Lakes Water Quality Board, International Joint Commission, Windsor, Ontario.

Gunnerson, C.G. 1988. Environmental issues for 21st Century engineers. Journal of Professional Issues in Engineering 114:287-301.

Haffner, G.D., Harris, G.P., and Jarai, M.K. 1980. Physical variability and phytoplankton communities. III. Vertical structure in phytoplankton populations. Archiv fur Hydrobiologie 89:363-381.

Halfon, E. 1981. Error analysis and simulation of mirex behaviour in Lake Ontario. Ecological Modelling 22:213-252.

Hamilton Harbour RAP Team. 1989. Remedial Action Plan for Hamilton Harbour. Stage I Report: Environmental conditions and problem definitions. Ontario Ministry of the Environment, Central Region, Hamilton, Ontario, 162 pp.

Harshbarger, J.C., and Clark, J.B. 1990. Epizootiology of neoplasms in bony fish of North America. Science of the Total Environment 94:1-32.

Hart, D.R., McKee, P.M., Burt, A.J., and Goffin, M.J. 1986. Benthic community and sediment quality assessment of Port Hope Harbour, Lake Ontario. Journal of Great Lakes Research 12:206-220.

Hayes, M.A., Smith, I.R., and Ferguson, H.W. 1987. Pathogenesis of neoplastic diseases afflicting feral fish. *In* Proceedings of the Technology Transfer Conference, November 30-December 1, 1987, Part D, Analytical Methods. Ontario Ministry of the Environment, Research Advisory Committee, Toronto, 15 pp.

Hayes, M.A., Smith, I.R., Rushmore, T.H., Crane, T.L., Thorn, C., Kocal, T.E., and Ferguson, H.W. 1990. Pathogenesis of skin and liver neoplasms in white suckers from industrially polluted areas in Lake Ontario. Science of the Total Environment 94:105-123.

Haynes, J.M., Nettles, D.C., Parnell, K.M., Voiland, M.P., Olson, R.A., and Winter, J.D. 1986. Movements of rainbow trout (*Salmo gairdneri*) in Lake Ontario and a hypothesis for the influence of spring thermal structure. Journal of Great Lakes Research 12:304-313.

Health and Welfare Canada. 1981. Tapwater Consumption in Canada. Health and Welfare Canada, Ottawa, Ontario.

Health and Welfare Canada. 1989. Guidelines for Canadian Drinking Water Quality, Fourth Edition. Health and Welfare Canada, Ottawa, Ontario, 25 pp.

Heaton, S.N. Aulerich, R.J., Bursian, S.J., Geisey, J.P., Rendes, J.A., Tillit, D.E. and Kubiak, J.J. 1991. Reproductive effects of feeding Saginaw Bay source fish to ranch mink. Cause-Effect Linkages II Symposium. Sept. 27-28, 1991, Grant Traverse Resort, Traverse City, Michigan.

Hill, H.A. 1965. The environment and disease: association or causation? Proceedings of the Royal Society of Medicine 58:295-300.

Hites, R.A., and Budde, W.L. 1991. EPA's analytical methods for water: the next generation. Environmental Science and Technology 25:998-1006.

Holdrinet, M.V.H., Frank, R., Thomas, R.L., Hetling, L.J. 1978. Mirex in the sediments of Lake Ontario. Journal of Great Lakes Research 4:69-74.

Holmes, J.A. 1986. The impact of dredging and spoils disposal on Hamilton Harbour fisheries: implications for rehabilitation. Canadian Technical Report of Fisheries and Aquatic Sciences, No. 1498, ix + 146 pp.

Holmes, J.A. 1988. Potential for fisheries rehabilitation in the Hamilton Harbour-Cootes Paradise ecosystem of Lake Ontario. Journal of Great Lakes Research 14:131-141.

Hornshaw, T.C., Aulerich, R.J., and Johnson, H.E. 1983. Feeding Great Lakes fish to mink: effects on mink and accumulation and elimination of PCBs by mink. Journal of Toxicology and Environmental Health 11:933-946.

Hurley, D.A. 1986. Fish populations in the Bay of Quinte, Lake Ontario, before and after phosphorus control. *In* Minns, C.K., Hurley, D.A., and Nicholls, K.H., [ed.], Project Quinte: point-source phosphorus control and ecosystem response in the Bay of Quinte, Lake Ontario. Canadian Special Publication of Fisheries and Aquatic Sciences, 86, pp. 201-214.

Hurley, D.A., and Christie, W.J. 1977. Depreciation of the warmwater fish community in the Bay of Quinte, Lake Ontario. Journal of the Fisheries Research Board of Canada 34:1849-1860.

Insalaco, S.E., Makarewicz, J.C., and McNamara, J.N. 1982. The influence of sex, size, and season on mirex levels within selected tissues of Lake Ontario salmon. Journal of Great Lakes Research 8:660-665.

Integrated Exploration. 1984. A Nearshore Benthic Survey of Lake Ontario. Integrated Exploration Report PJ8102, Submitted to the Ontario Ministry of Environment, Water Resources Branch, Toronto, Ontario, 60 pp.

International Joint Commission. 1989. Great Lakes Water Quality Agreement of 1978, as amended by Protocol signed November 18, 1987. Windsor, Ontario, 84 pp.

International Joint Commission. 1991. Commission approves list/delist criteria for Great Lakes Areas of Concern. Focus 16(1):1-5.

Jaffe, R., and Hites, R.A. 1986. Anthropogenic, polyhalogenated, organic compounds in non-migratory fish from the Niagara River area and tributaries to Lake Ontario. Journal of Great Lakes Research 12:63-71.

Jardine, C.G., and Bowman, A.B. 1990. Spanish River Remedial Action Plan, Fish Tainting Evaluation. Spanish Harbour Remedial Action Plan, Ontario Ministry of the Environment, Northeastern Region, Technical Report SR-90-01, 15 pp.

Johannsson, O.E., Dermott, R.M., Feldkamp, R., and Moore, J.E. 1985. Lake Ontario Long-Term Monitoring Biological Program: Report for 1981 and 1982. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1414, xvi + 208 pp.

Johannsson, O.E., Mills, E.L., and O'Gorman, R. 1991. Changes in the nearshore and offshore zooplankton communities in Lake Ontario: 1981-1988. Canadian Journal of Fisheries and Aquatic Sciences 48:1546-1557.

Johnson, A.F., Cox, C.M., and Vaillancourt, A.L. 1989. Contaminants in Ontario sport fish - long term trends and future prospects. *In* Proceedings Environmental Research: 1989 Technology Transfer Conference, Volume 1. Feature Presentations, Air Quality Research, Water Quality Research, Novemeber 20-21, 1989. Ontario Ministry of the Environment, Research and Technology Branch, Toronto, Ontario, pp. 285-293.

Kaminsky, R., Kaiser, K.L.E., and Hites, R.A. 1983. Fates of organic compounds from Niagara Falls dumpsites in Lake Ontario. Journal of Great Lakes Research 9:183-189.

Kaminsky, R., and Hites, R.A. 1984. Octachlorostyrene in Lake Ontario: Sources and fates. Environmental Science and Technology 18:275-279.

Kirby, G.M., Smith, I.R., Thorn, C., Ferguson, H.W., and Hayes, M.A. 1988. Potential role of polycyclic aromatic hydrocarbons in the development of liver tumors in fish from polluted sites of Lake Ontario. *In* Proceedings of the Technology Transfer Conference 1988. Session B, Water Quality Research. Ontario Ministry of the Environment, Research and Technology Branch, Toronto, Ontario, pp. 55-65.

Kirby, G.M., Smith, I.R., Bend, J.R., Ferguson, H.W., and Hayes, M.A. 1989a. Importance of glutathione s-transferase and concurrent hepatic disease in the development of liver neoplasms in white suckers (*Catostomus commersoni*) from industrially polluted regions of Lake Ontario. *In* Proceedings, Environmental Research: 1989 Technology Transfer Conference, Volume 1. Ontario Ministry of the Environment, Research and Technology Branch, Toronto, Ontario, pp. 311-320.

Kirby, G.M., Smith, I.R., and Hayes, M.A. 1989b. Pharmacokinetics of ^{3}H-benzo(*a*)pyrene administered to white suckers from polluted and reference sites in the Great Lakes. *In* Van Coillie, R., Niimi, A., Champoux, A., and Joubert, G., [ed.], Proceedings of the Fifteenth Annual Aquatic Toxicity Workshop: November 28-30, 1988, Montreal, Quebec. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1714.

Kuntz, K. 1988. Recent Trends in Water Quality of the Niagara River. Environment Canada, Inland Waters Directorate, Burlington, Technical Bulletin No. 146, v + 15 pp.

Kwiatkowski, R.E., and Neilson, M.A.T. 1983. Lake Ontario Surveillance Data 1968-1980. Environment Canada, Inland Waters Directorate, Burlington, Technical Bulletin No. 126, vii + 69 pp.

Lake Ontario Secretariat. 1990. Lake Ontario Toxics Management Plan (draft update). Environment Canada, U.S. Environment Protection Agency, Ontario Ministry of the Environment, and New York State Department of Environmental Conservation, 400 pp.

Lake Ontario Toxics Committee. 1989. Lake Ontario Toxics Management Plan. A Report by the Lake Ontario Toxics Committee: Environment Canada, United States Environmental Protection Agency, Ontario Ministry of the Environment and New York State Department of Environmental Conservation.

Lean, D.R.S. 1987. Nutrient status of Lake Ontario. Canadian Journal of Fisheries and Aquatic Sciences 44:2042-2046.

Lean, D.R.S., Frick, H.-J., Charlton, M.N., Cuhel, R.L. and Pick, F.R. 1987. The Lake Ontario life support system. Canadian Journal of Fish and Aquatic Sciences 44:2230-2240.

Leatherland, J.F., and Sonstegard, R.A. 1980. Structure of thyroid and adrenal glands in rats fed diets of Great Lakes coho salmon (*Oncorhynchus kisutch*). Environmental Research 23:77-86.

Leonardelli, S. 1990. Evaluation of Phosphorus Control Programs in Ontario. Environment Canada, Ontario Region, Environmental Protection Service, manuscript, 47 pp.

LeTendre, G.C., and Savoie, P.J. 1990. Lake Ontario Stocking and Marking Program for 1989, Chapter 11. *In* Lake Ontario Fisheries Unit, 1989 Annual Report to the Lake Ontario Committee of the Great Lakes Fishery Commission. Ontario Ministry of Natural Resources, Picton, Ontario, Report LOA 90.1.

Levings, C.D. 1982. The Ecological Consequences of Dredging and Dredge Spoil Disposal in Canadian Waters. National Research Council of Canada, Ottawa, NRCC Publication No. 18130, 142 pp.

Likens, G.E., [ed.]. 1972. Nutrients and eutrophication: the limiting nutrient controversy. Proceedings of Symposium, W.K.Kellogg Biological Station, Michigan State University, 1971, Special Symposium Volume 1: American Society of Limnology and Oceanography.

Lum, K.R., Kaiser, K.L.E, and Comba, M.E. 1987. Export of mirex from Lake Ontario to the St. Lawrence Estuary. Science of the Total Environment 67:41-51.

Luxon, P.L., Hodson, P.V., and Borgmann, U. 1987. Hepatic aryl hydrocarbon hydroxylase activity of lake trout (*Salvelinus namaycush*) as an indicator of organic pollution. Environmental Toxicology and Chemistry 6:649-657.

Mac, M.J. 1990. Lake trout egg quality in Lakes Michigan and Ontario. *In* Proceedings of the expert consultation on contaminant caused reproductive problems in salmonids. Report of the Ecological Committee, Science Advisory Board, to the International Joint Commission, Windsor, Ontario.

Mac, M.J., and Edsall, C.C. 1991. Environmental contaminants and the reproductive success of lake trout in the Great Lakes: an epidemiological approach. Journal of Toxicology and Environmental Health (in press).

Mac, M.J., Edsall, C.C., and Seelye, J.G. 1985. Survival of lake trout eggs and fry reared in water from the upper Great Lakes. Journal of Great Lakes Research 11:520-529.

Maccubbin, A.E., Chidambaram, S., and Black, J.J. 1988. Metabolites of aromatic hydrocarbons in the bile of brown bullheads (*Ictalurus nebulosus*). Journal of Great Lakes Research 14:101-108.

Makarewicz, J.C. 1985. Phytoplankton composition, abundance and distribution: Oswego River and Harbor and Niagara River plume. *In* Bertram, P.E., [ed.], Limnology and phytoplankton structure in nearshore areas of Lake Ontario: 1981. U.S. Environmental Protection Agency, Great Lakes National Program Office, Chicago, Illinois, EPA-905/3-85-003, pp. 97-172.

Marcquenski, S. 1990. Drop-out syndrome in steelhead fry in Wisconsin. *In* Proceedings of the expert consultation on contaminant caused reproductive problems in salmonids. Report of the Ecological Committee, Science Advisory Board, to the International Joint Commission, Windsor, Ontario.

Marsden, J.E., Krueger, C.C., and Schneider, C.P. 1988. Evidence of natural reproduction by stocked lake trout in Lake Ontario. Journal of Great Lakes Research 14:3-8.

Metcalfe, C.D., Cairns, V.W., and Fitzsimons, J.D. 1988. Experimental induction of liver tumours in rainbow trout (*Salmo gairdneri*) by contaminated sediment from Hamilton Harbour, Ontario. Canadian Journal of Fisheries and Aquatic Sciences 45:2161-2167.

Metcalfe, C.D., Balch, G.C., Cairns, V.W., Fitzsimons, J.D., and Dunn, B.P. 1990. Carcinogenic and genotoxic activity of extracts from contaminated sediments in western Lake Ontario. Science of the Total Environment 94:125-141.

Metcalfe, J.L., Fox, M.E., and Carey, J.H. 1988. Freshwater leeches (Hirundinea) as a screening tool for detecting organic contaminants in the environment. Environmental Monitoring and Assessment 11:147-169.

Metro Toronto RAP Team. 1988. Metro Toronto Remedial Action Plan. Stage I. Environmental conditions and problem definition. Ontario Ministry of the Environment, Toronto, Ontario, 196 pp + appendices.

Michigan Department of Natural Resources. 1980. Critical Materials Register. Michigan Department of Natural Resources, Environmental Services Division, Lansing, Michigan.

Millner, G.A.. and Sweeney, R.A. 1982. Lake Erie *Cladophora* in perspective. Journal of Great Lakes Research 8:27-29.

Moccia, R.D., Fox, G.A., and Britton, A. 1986. A quantitative assessment of thyroid histopathology of herring gulls (*Larus argentatus*) from the Great Lakes and a hypothesis on the causal role of environmental contaminants. Journal of Wildlife Diseases 22:60-70.

Moccia, R.D., Leatherland, J.F., and Sonstegard, R.A. 1977. Increasing frequency of thyroid goiters in coho salmon (*Oncorhynchus kisutch*) in the Great Lakes. Science 198(4315):425-426.

Morris, R.D., Hunter, R.A., and McElman, J.F. 1976. Factors affecting the reproductive success of common tern (*Sterna hirundo*) colonies on the lower Great Lakes during the summer of 1972. Canadian Journal of Zoology 54:1850-1862.

Morris, R.D., Kirkham, I.R., and Chardine, J.W. 1980. Management of a declining common tern colony. Journal of Wildlife Management 44:241-245.

Mudroch, A. 1983. Distribution of major elements and metals in sediment cores from the western basin of Lake Ontario. Journal of Great Lakes Research 9:125-133.

Mudroch, A., and Duncan, G.A. 1986. Distribution of metals in different size fractions of sediment from the Niagara River. Journal of Great Lakes Research 12:117-126.

Mudroch, A., Sarazin, L., and Lomas, T. 1988. Summary of surface and background concentrations of selected elements in the Great Lakes sediments. Journal of Great Lakes Research 14:241-251.

Munawar, M., and Thomas, R.L. 1989. Sediment toxicity testing in two areas of concern of the Laurentian Great Lakes: Toronto (Ontario) and Toledo (Ohio) Harbours. Hydrobiologia 176/177:397-409.

Munawar, M., and Weisse, T. 1989. Is the 'microbial loop' an early warning indicator of anthropogenic stress? Hydrobiologia 188/189:163-174.

Munawar, M., Munawar, I.F., and McCarthy, L.H. 1987. Phytoplankton ecology of large eutrophic and oligotrophic lakes of North America: Lakes Ontario and Superior. Archiv für Hydrol biologia Beih. Ergebn. Limnologische 25:51-96.

Munawar, M., Munawar, I.F., Mayfield, C.I., and McCarthy, L.H. 1989. Probing ecosystem health: a multi-disciplinary and multi-trophic assay strategy. Hydrobiologia 188/189:93-116.

Nalepa, T.F. 1991. Status and trends of the Lake Ontario macrobenthos. Canadian Journal of Fisheries and Aquatic Sciences 48:1558-1567.

Nalewajko, C., Ewing, C., and Mudroch, C. 1989a. A comparision of the effects of sediments and standard elutriates on phosphorus kinetics in lakewater. Water, Air and Soil Pollution 44:119-134.

Nalewajko, C., Ewing, C., and Mudroch, A. 1989b. Effects of contaminated sediments and standard elutriates on primary production in Lake Ontario. Water, Air and Soil Pollution 45:275-286.

National Wildlife Federation and the Canadian Insitute for Environmental Law and Policy. 1991. A prescription for healthy Great Lakes. Report of the Program for Zero Discharge. NWF/CIELP, Ann Arbor, Michigan, and Toronto, Ontario, 64 pp.

National Research Council of the United States and the Royal Society of Canada. 1985. The Great Lakes Water Quality Agreement: An evolving instrument for ecosystem management. National Academy Press, Washington, D.C.

Neil, J.H., and Jackson, M.B. 1982. Monitoring *Cladophora* growth and the effect of phosphorus additions at a shoreline site in northeastern Lake Erie. Journal of Great Lakes Research 8:30-34.

Neilson, M.A., and Stevens, R.J.J. 1987. Spatial heterogeneity of nutrients and organic matter in Lake Ontario. Canadian Journal of Fisheries and Aquatic Sciences 44:2192-2203.

New York State Department of Environmental Conservation. 1987. 1.1.1 Ambient Water Quality Standards and Guidance Values. *In* Technical Operating Guidance Series, Division of Water, New York State Department of Environmental Conservation, Albany, New York.

New York State Department of Environmental Conservation. 1990. Oswego River Remedial Action Plan. Stage I. New York State Department of Environmental Conservation and Oswego River Citizens' Advisory Committee, Oswego, New York, 114 pp.

New York State Department of Environmental Conservation. 1991. 1991 Annual Report of Bureau of Fisheries, Great Lakes Fisheries Section, Lake Ontario Unit to the Lake Ontario Committee and the Great Lakes Fishery Commission. Division of Fish and Wildlife, Bureau of Fisheries, Lake Ontario Unit, Albany, New York, 152 pp.

New York State Department of Health. 1990. Health Advisory. Chemicals in Sportfish or Game 1990-1991. New York State Department of Health, Division of Environmental Health Assessment, Albany, New York, 8 pp.

Newell, A.J., Johnson, D.W., and Allen, L.K. 1987. Niagara River Biota Contamination Project: Fish Flesh Criteria for Piscivorous Wildlife. New York State Department of Environmental Conservation, Division of Fish and Wildlife, Bureau of Environmental Protection, Albany, New York, Technical Report 87-3, 182 pp.

Niagara River RAP Team. 1990. Final Draft. Remedial Action Plan for the Niagara River (Ontario) Area of Concern. Stage 1: Environmental conditions and problem definition. Ontario Ministry of the Environment, Toronto, Ontario, 177 pp.

Niagara River Toxics Committee. 1984. Report of the Niagara River Toxics Committee. U.S. Environment Protection Agency, Ontario Ministry of the Environment, Environment Canada, and New York State Department of Environmental Conservation, 630 pp.

Nieboer, E., and Jusys, A.A. 1983. Contamination control in routine ultratrace analysis of toxic metals. *In* Brown, S.S. and Savory, H. [eds.], Chemical Toxicology and Clinical Chemistry of Metals. Academic Press, London, pp. 3-16.

Niimi, A.J., and Oliver, B.G. 1989. Distribution of polychlorinated biphenyl congeners and other halocarbons in whole fish and muscle among Lake Ontario salmonids. Environmental Science and Technology 23:83-88.

Norstrom, R.J., Braune, B.M., Macdonald, C.R., Simon, M., and Weseloh, D.V. 1989. Levels and trends of PCDDs and PCDFs in Great Lakes herring gull eggs, 1981-1988. Poster, Dioxin 89, SETAC, September 17-22, 1989, Toronto, Ontario.

Nriagu, J.O., Wong, H.K.T., and Snodgrass, W.J. 1983. Historical records of metal pollution in sediments of Toronto and Hamilton Harbours. Journal of Great Lakes Research 9:365-373.

NYCRR 1990. Chapter 10, New York Compilation of Rules and Regulations, Part 5. Public Water Systems, New York State.

O'Gorman, R., Bergstedt, R.A., and Eckert, T.H. 1987. Prey fish dynamics and salmonine predator growth in Lake Ontario, 1978-84. Canadian Journal of Fisheries and Aquatic Sciences 44(Supplement 2):390-403.

Organization for Economic Co-operation and Development. 1981. Guidelines for Testing of Chemicals. OECD Publications, Paris, France.

Oliver, B.G., and Nichol, K.D. 1982. Chlorobenzenes in sediments, water and selected fish from Lakes Superior, Huron, Erie and Ontario. Environmental Science and Technology 16:532-536.

Oliver, B., and Charlton, M.N. 1984. Chlorinated organic contaminants on settling particulates in the Niagara River vicinity of Lake Ontario. Environmental Science and Technology 18:903-908.

Ontario Ministry of the Environment. 1978. Water Management: Goals, Policies, Objectives and Implementation Procedures of the Ministry of the Environment. Toronto, Ontario.

Ontario Ministry of the Environment. 1983. Ontario Drinking Water Objectives. Ontario Ministry of the Environment. Toronto, Ontario, 56 pp.

Ontario Ministry of the Environment. 1986. Municipal-Industrial Strategy for Abatement (MISA): A Policy and Program Statement of the Government of Ontario on Controlling Municipal and Industrial Discharges into Surface Waters, Ontario Ministry of the Environment. Toronto, Ontario, 53 pp.

Ontario Ministry of the Environment. 1988a. MISA: Cost Estimates and Implications of the Effluent Monitoring - General and Effluent Monitoring - Petroleum Refining Section, Regulations for Ontario's Petroleum Refineries, Ontario Ministry of the Environment. Toronto, Ontario.

Ontario Ministry of the Environment. 1988b. News Release. Study confirms U.S. sources of dioxin, PCBs and mirex in Niagara River. Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario, May 31, 1988, 6 pp.

Ontario Ministry of the Environment. 1989a. MISA: Monitoring Costs and Their Implications for Direct Dischargers in the Ontario Mineral Industry: Group "B" Industrial Minerals Sector. Ontario Ministry of the Environment, Toronto, Ontario.

Ontario Ministry of the Environment. 1989b. MISA: Monitoring Costs and Their Implications for Direct Dischargers in the Ontario Inorganic Chemical Industry - Final Report, Ontario Ministry of the Environment, Toronto, Ontario.

Ontario Ministry of the Environment. 1989c. Drinking Water Surveillance Program Overview Annual Report. Ontario Ministry of the Environment, Toronto, Ontario, 180 pp.

Ontario Ministry of the Environment. 1990a. Ontario Drinking Water Objectives, Draft Revision 1990. ISBN 0-7743-8985-0.

Ontario Ministry of the Environment. 1990b. St. Catharines Area Pollution Control Plan (SCAPCP): Final Report, Pollution Control Strategy. Ontario Ministry of the Environment, Toronto, Ontario, 182 pp.

Ontario Ministry of the Environment. 1991. Parameters Listing System (PALIS). Ontario Ministry of the Environment, Toronto, Ontario, 80 pp.

Ontario Ministry of the Environment and Ministry of Natural Resources. 1991. Guide to eating Ontario sport fish. 1991. Fifteenth edition, revised. Toronto, Ontario, 180 pp.

Ontario Ministry of Health. 1975. Water Quality Guidelines for Bathing Beaches. Ontario Ministry of Health, Toronto, Ontario.

Ontario Ministry of Natural Resources. 1991. 1990 Annual Report of the Lake Ontario Fisheries Unit. Lake Ontario Fisheries Unit, Picton, Ontario, 187 pp.

Painter, D.S., and Kamaitis, G. 1985. Reduction in *Cladophora* biomass and tissue phosphorus concentration in Lake Ontario, 1972- 1983. Canadian Journal of Fisheries and Aquatic Sciences 44:2212-2215.

Palmateer, G.A., Dutka, B.J., Janzen, E.M., Meissner, S.M., and Sakellaris, M. 1990. Coliphages and bacteriophages in Canadian drinking waters. Water International 15:157-159.

Peakall, D.B., and Fox, G.A. 1987. Toxicological investigations of pollutant-related effects in Great Lakes gulls. Environmental Health Perspectives 71:187-193.

Peakall, D.B., Fox, G.A., Gilman, A.P., Hallett, D.J., and Norstrom, R.J. 1980. Reproductive success of herring gulls as an indicator of Great Lakes water quality. *In* Afghan, D.K., and Mackay, D., [ed.], Hydrocarbons and Halogenated Hydrocarbons in the Aquatic Environment. Plenum Press, New York, pp. 337-344.

Peet, M.E. 1990. Water Quality Management Process and Progress 1987-1990. Monroe County, New York, 30 pp.

Persaud, D., and Wilkins, W.D. 1976. Evaluating Construction Activities Impacting on Water Resources. Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario, 76 pp.

Persaud, D., Jaagumagi, R., and Hayton, A. 1991. The Provincial Sediment Quality Guidelines. Draft. Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario, iii + 26 pp.

Port Hope RAP Team. 1989. Port Hope Harbour Remedial Action Plan. Stage I. Environmental conditions and problem definition. Ontario Ministry of the Environment, Toronto, Ontario, viii + 95 pp + appendices.

Price, I.M., and Weseloh, D.V. 1986. Increased numbers and productivity of double-crested cormorants, *Phalacrocorax auritus*, on Lake Ontario. Canadian Field-Naturalist 100:474-482.

Rasmussen, J.B., Rowan, D.J., Lean, D.R.S., and Carey, J.H. 1990. Food chain structure in Ontario lakes determines PCB levels in lake trout (*Salvelinus namaycush*) and other pelagic fish. Canadian Journal of Fisheries and Aquatic Sciences 47:2030-2038.

Rattner, B., Hoffman, D.J., and Marn, C.M. 1989. Use of mixed-function oxygenases to monitor contaminant exposure in wildlife. Environmental Toxicology and Chemistry 8:1093-1102.

Rodgers, G.K. 1987. Time of onset of full thermal stratification in Lake Ontario in relation to lake temperatures in winter. Canadian Journal of Fisheries and Aquatic Sciences 44:2225-2229.

Rosa, F. 1985. Sedimentation and sediment resuspension in Lake Ontario. Journal of Great Lakes Research 11:13-25.

Rosa, F., Nriagu, J.O., and Wong, H.K. 1983. Particulate flux at the bottom of Lake Ontario. Chemosphere 12:1345-1354.

Royal Commission on the Future of the Toronto Waterfront. 1989a. Environment and Health. Royal Commission on the Future of the Toronto Waterfront. Toronto, Ontario, 151 pp.

Royal Commission on the Future of the Toronto Waterfront. 1989b. Parks, Pleasures, and Public Amenities. Royal Commission on the Future of the Toronto Waterfront. Toronto, Ontario, 129 pp.

Royal Commission on the Future of the Toronto Waterfront. 1990. Watershed. Interim Report August 1990. Royal Commission on the Future of the Toronto Waterfront. Toronto, Ontario, 207 pp.

Ruby, S.M., and Cairns, V.W. 1983. Implications of testicular constrictions on spermatogenesis in Lake Ontario lake trout, *Salvelinus namaycush*. Journal of Fish Biology 23:385-395.

Ryder, R.A., and Edwards, C.J., [ed.]. 1985. A conceptual approach for the application of biological indicators of ecosystem quality in the Great Lakes basin. A Joint Effort of the International Joint Commission and the Great Lakes Fishery Commission. Report to the Science Advisory Board, International Joint Commission, Windsor, Ontario, 169 pp.

Sackett, D.L. 1980. A review of diagnostic tests for causation. *In* Ontario Royal Commission on Asbestos, Proceedings, Second Public Meeting, Friday, December 12, 1980, pp. 10-16.

Safe, S. 1990. Polychlorinated biphenyls (PCBs), dibenzo-p-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: environmental and mechanistic considerations which support the development of toxic equivalency factors (TEFs). Critical Reviews in Toxicology 21:51-88.

Schindler, D.W. 1971. Evolution of phosphorus limitation in lakes. Science 195:260-262.

Schneider, C.P., Eckert, T.H., O'Gorman, R., Elrod, J.H., Owens, R.W., Schaner, T., Grewe, P., Marsden, J.E., and Krueger, C.C. 1990. Surveillance Report for Lake Trout Rehabilitation in Lake Ontario, 1990, Chapter 8. *In* Lake Ontario Fisheries Unit, 1989 Annual Report to the Lake Ontario Committee, Great Lakes Fishery Commission. Ontario Ministry of Natural Resources, Piction, Report LOA 90.1.

Schneider, C.P., Kolenosky, D.P., and Goldthwaite, D.B. 1983. A Joint Plan for the Rehabilitation of Lake Trout in Lake Ontario. Great Lakes Fishery Commission, Lake Ontario Committee, Special Publication, 50 pp.

Sediment Subcommittee. 1988a. Procedures for the Assessment of Contaminated Sediment Problems in the Great Lakes. Report to the Great Lakes Water Quality Board, International Joint Commission, Windsor, Ontario, ix + 140 pp.

Sediment Subcommittee. 1988b. Options for the Remediation of Contaminated Sediments in the Great Lakes. Report to the Great Lakes Water Quality Board, International Joint Commission, Windsor, Ontario, ix + 78 pp.

Sediment Work Group. 1990. Register of Great Lakes Dredging Projects 1980-1984. Report to the Great Lakes Water Quality Board, International Joint Commission, Windsor, Ontario, vi + 203 pp.

Sediment Work Group. 1991. Register of Great Lakes Dredging Projects 1985-1989. Report to the Great Lakes Water Quality Board, International Joint Commission, Windsor, Ontario, (Draft).

Sileo, L., Karstad, L., Frank, R., Holdrinet, M.V.H., Addison, E., and Braun, H.E. 1977. Organochlorine poisoning of ring-billed gulls in southern Ontario. Journal of Wildlife Diseases 13:313-321.

Simonin, H., Skea, J., Dean, H., and Symula, J. 1990. Summary of reproductive studies of Lake Ontario salmonids. *In* Proceedings of the expert consultation on contaminant caused reproductive problems in salmonids. Report of the Ecological Committee, Science Advisory Board, to the International Joint Commission, Windsor, Ontario.

Skea, J.C., Simonin, H.A., Harris, E.J., Jackling, S., Spagnoli, J.J., Symula, J., and Colquhoun, J.R. 1979. Reducing levels of mirex, Aroclor 1254, and DDE by trimming and cooking Lake Ontrio brown trout (*Salmo trutta* Linnaeus) and smallmouth bass (*Micropterus dolomieui* Lacepede). Journal of Great Lakes Research 5:153-159.

Skea, J.C., Symula, J., and Miccoli, J. 1985. Separating starvation losses from other early feeding fry mortality in steelhead trout *Salmo gairdneri*, chinook salmon *Oncorhynchus tshawaytscha*, and lake trout *Salvelinus namaycush*. Bulletin of Environmental Contamination and Toxicology 35:82-91.

Skinner, L.C. 1991. Unpublished Lake Ontario fish contaminant data. Presented at the Lake Ontario Committee Meeting of the Great Lakes Fishery Commission, 27-28 March 1991, Niagara Falls, New York, 7 pp.

Skinner, L.C., and Jackling, S.J. 1989. Chemical contaminants in young-of-the-year fish from New York's Great Lakes basin: 1984 through 1987. New York State Department of Environmental Conservation, Division of Fish and Wildlife, Bureau of Environmental Protection, Albany, New York, Technical Report 89-1 (BEP), 41 pp.

Sleigh, J.D., and Timburg, M.C. 1986. Notes on Medical Bacteriology, 2nd Edition, Churchill Livingstone, New York.

Sloan, R., with O'Connell, E., and Diana, R. 1987. Toxic Substances in Fish and Wildlife: Analyses Since May 1, 1982. Volume 6. New York State Department of Environmental Conservation, Division of Fish and Wildlife, Bureau of Environmental Protection, Albany, New York, Technical Report 87-4 (BEP), 182 pp.

Sly, P.G. 1984. Sedimentation and geochemistry of modern sediments in the Kingston Basin of Lake Ontario. Journal of Great Lakes Research 10:358-374.

Sly, P.G. 1988. Interstitial water quality of lake trout spawning habitat. Journal of Great Lakes Research 14:301-315.

Sly, P.G. 1991. The effects of land use and cultural development on the Lake Ontario ecosystem since 1750. Hydrobiologia 213:1-75.

Smardon, R. 1989. Human Perception of Utilization of Wetlands for Waste Assimilation, or How Do You Make a Silk Purse Out of a Sow's Ear. *In* Hammer, D.A. [ed.], Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural. Lewis Publishers Inc., Michigan, pp. 287-295.

Smith, I.R., and Ferguson, H.W. 1985. The assessment of a point source discharge of suspected mutagenic and carcinogenic contaminants: an epidemiological approach. *In* Proceedings of the Technology Transfer Conference No. 6 (Part 2), Ontario Ministry of the Environment, pp. 285-332.

Smith, I.R., and Zajdlik, B.A. 1987. Regression and development of epidermal papillomas affecting white suckers, *Catostomus commersoni* (Lacepede), from Lake Ontario, Canada. Journal of Fish Diseases 10:487-494.

Smith, I.R., Ferguson, H.W., and Hayes, M.A. 1989a. Histopathology and prevalence of epidermal papillomas epidemic in brown bullhead, *Ictalurus nebulosus* (Lesueur), and white sucker, *Catostomus commersoni* (Lacepede), populations from Ontario, Canada. Journal of Fish Diseases 12:373-388.

Smith, I.R., Zajdlik, B.A., Ferguson, H.W., and Hayes, M.A. 1989b. Regression and development of skin papillomas affecting white suckers (*Catostomus commersoni*) from polluted areas in Lake Ontario. Abstract only. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1607, pp. 177.

Smith, I.R., Marchant, B., and Frimeth, J. 1991. Lake Ontario Pacific salmon reproduction. Presented at the Lake Ontario Committee Meeting of the Great Lakes Fishery Commission, March 27-28, 1991, Niagara Falls, New York.

Sonstegard, R.A. 1977. Environmental carcinogenesis studies in fishes of the Great Lakes of North America. Annals of the New York Academy of Sciences 298:261-269.

Sonstegard, R.A., and Leatherland, J.F. 1979. Hypothyroidism in rats fed Great Lakes coho salmon. Bulletin of Environmental Contamination and Toxicology 22:779-784.

Spear, P.A., Bourbonnais, D.H., Norstrom, R.J., and Moon, T.W. 1990. Yolk retinoids (vitamin A) in eggs of the herring gull and correlations with polychlorinated-p-dioxins and dibenzofurans. Environmental Toxicology and Chemistry 9:1053-1061.

Spear, P.A., Moon, T.W., and Peakall, D.B. 1986. Liver retinoid concentrations in natural populations of herring gulls (*Larus argentatus*) contaminated by 2,3,7,8-tetrachlorodibenzo-p-dioxin and in ring doves (*Streptopelia risoria*) injected with a dioxin analogue. Canadian Journal of Zoology 64:204-208.

Sreenivasa, M.R., and Nalewajko, C. 1975. Phytoplankton biomass and species composition in northeastern Lake Ontario, April-November, 1965. Journal of Great Lakes Research 1:151-161.

Stevens, R.J.J. 1988. A Review of Lake Ontario Water Quality With Emphasis on the 1981-1982 Intensive Years. Report to the Surveillance Subcommittee, Great Lakes Water Quality Board, International Joint Commission, Windsor, Ontario, xvi + 300 pp.

Stevens, R.J.J., and Neilson, M.A.T. 1987. Response of Lake Ontario to reductions in phosphorus load, 1967-1982. Canadian Journal of Fisheries and Aquatic Sciences 44:2059-2068.

Struger, J., Elliott, J.E., and Weseloh, D.V. 1987. Metals and essential elements in herring gulls from the Great Lakes, 1983. Journal of Great Lakes Research 13:43-55.

Suns, K., Hitchin, G., and Adamek, E. 1991. Present status and temporal trends of organochlorine contaminants in young-of-the-year spottail shiners (*Notropis hudsonius*) from Lake Ontario. Canadian Journal of Fisheries and Aquatic Sciences 48:1568-1573.

Svenson, R.W. 1990. Testimony Before the Subcommittee on Superfund. Ocean and Water Protection Committee on Environment and Public Works, United States Senate, 4 pp.

Symula, J., Meade, J., Skea, J.C., Cummings, L., Colquhoun, J.R., Dean, H.J., and Miccoli, J. 1990. Blue-sac disease in Lake Ontario lake trout. Journal of Great Lakes Research 16:41-52.

Taylor, S.M., Frank, J., Haight, M., Streiner, D., Walter, S., White, N., and Willms, D. 1988. The psychosocial impacts of exposure to environmental contaminants in Ontario: a feasibility study. *In* Proceedings Technology Transfer Conference 1988, Session E, Environmental Economics. Ontario Ministry of the Environment, Toronto, Ontario, pp. 37-49.

Teeple, S.M. 1977. Reproductive success of herring gulls nesting on Brothers Island, Lake Ontario, in 1973. Canadian Field-Naturalist 91:148-157.

Thomas, R.L. 1983. Lake Ontario sediments as indicators of the Niagara River as a primary source of contaminants. Journal of Great Lakes Research 9:118-124.

Toronto Department of Public Health. 1991. Six Year Summary of Beach Postings (1985-1990), Department of Public Health, Toronto, Ontario.

Toth, C. 1990. St. Catharines Area Pollution Control Plan (SCAPCP): St. Catharines Area Beaches 1988, 67 pp.

Usher, A., Ellis, J.B., and Michalski, M. 1987. Beach Use and Environmental Quality in Ontario. Report prepared for the Ontario Ministry of the Environment, Toronto, Ontario, 203 pp.

Virtual Elimination Task Force. 1991. Persistent Toxic Substances. Virtually eliminating inputs to the Great Lakes. Summary for Discussion Purposes, Prepared for the International Joint Commission, Windsor, Ontario, 23 pp.

Vollenweider, R.A. 1968. The scientific basis of lake and shear eutrophication, with particular reference to phosphorus and nitrogen as eutrophication factors. Organization for Economic Cooperation and Development, Paris, France, Technical Report DAS/CSI/68, 27:1-182.

Walker, M.K., Spitsbergen, J.M., Olson, J.R., and Peterson, R.E. 1991. 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin (TCDD) toxicity during early life stage development of lake trout (*Salvelinus namaycush*). Canadian Journal of Fisheries and Aquatic Sciences 48:875-883.

Wang, G.M., and Schwetz, B.A. 1987. An evaluation system for ranking chemicals with teratogenic potential. Teratogenesis, Carcinogenesis and Mutagenesis 7:133-139.

Warwick, W.F., Fitchko, J., McKee, P.M., Hart, D.R., and Burt, A.J. 1987. The incidence of deformities in *Chironomous* spp. from Port Hope Harbour, Lake Ontario. Journal of Great Lakes Research 13:88-92.

Weekes, F. 1974. A survey of bald eagle nesting attempts in southern Ontario, 1969-1973. Canadian Field-Naturalist 88:415-419.

Weekes, F.M. 1975. Bald eagle nesting attempts in southern Ontario in 1974. Canadian Field-Naturalist 89:438-444.

Welch, J. 1978. The Impact of Inorganic Phosphates on the Environment. United States Environmental Protection Agency, Office of Toxic Substances, Washington, D.C.

Weseloh, D.V. 1984. The origins of banded herring gulls recovered in the Great Lakes region. Journal of Field Ornithology 55:190-195.

Weseloh, D.V., Custer, T.W., and Braune, B.M. 1989. Organochlorine contaminants in eggs of common terns from the Canadian Great Lakes, 1981. Environmental Pollution 59:141-160.

Weseloh, D.V., Mineau, P, and Struger, J. 1990. Geographical distribution of contaminants and productivity measures of herring gulls in the Great Lakes: Lake Erie and connecting channels 1978/79. Science of the Total Environment 91:141-159.

Whillans, T.H. 1979. Historic transformations of fish communities in three Great Lakes bays. Journal of Great Lakes Research 5:195-215.

Whillans, T.H. 1982. Changes in marsh area along the Canadian shore of Lake Ontario. Journal of Great Lakes Research 8:570-577.

Whittle, D.M., and Keir, M.J. 1991. Spatial and temporal contaminant trends in Great Lakes lake trout and smelt (1986-1989). Unpublished manuscript, Department of Fisheries and Oceans, Great Lakes Laboratory for Fisheries and Aquatic Sciences, Burlington, Ontario, 12 pp.

Wilkins, W.D. 1985. Sediment Quality and Benthic Macroinvertebrates at 25 Transects in the Lake Ontario Nearshore Zone 1981. Manuscript report prepared for Ontario Ministry of Environment, Water Resources Branch, Toronto, 79 pp + appendices.

Wren, C.D. 1991. Cause-effect linkages between chemicals and populations of mink (*Mustela vison*) and otter (*Lutra canadensis*) in the Great Lakes basin. Journal of Toxicology and Environmental Health 33:549-585.

APPENDIX A

IMPAIRMENT OF BENEFICIAL USES STUDY: CONTACT LIST

Contact Name	Affiliation
Addison, Ed	MNR, Wildlife Branch, Maple
Aspenall, Doug	OMAF, Guelph
Baines, Chris	Centre for Great Lakes
Banister, Jeff	Molsons, Etobicoke
Barrett, Suzanne	Crombie Commission
Beltran, Bob	US EPA
Bertram, Paul	US EPA
Black, Jack	Roswell Park, Buffalo
Blokpoel, Hans	CWS, Ottawa
Boyd, Duncan	MOE
Boyer, Peter	IJC
Brindle, Ian	Brock University
Bryant, Doug	McMaster University
Burham, Bob	NYS Dept. of Health
Burke, Mike	NYS Health Dept.
Busch, Dieter	US FWS
Cairns, Vic	DFO, Burlington
Campbell, Monica	Environmental Protection Office, City of Toronto
Carreiro, Joe	CWS, Ottawa
Coape- Arnold, Tom	MOE, RAP coordinator
Colborn, Theo	World Wildlife Fund, Washington
Colucci, Rosemary	Board of Trade, Environment Committee
Coote, Dick	Agriculture Canada
Cork, Phil	US EPA, Duluth
Crowder, Adele	Queen's University
Darnell, Susan	Consultant to Agriculture Canada
Davis, Tony	Environment Canada
Dermott, Ron	DFO
Dickinson, T.	University of Guelph
Dickman, Mike	Brock University
Dobson, Hugh	Environment Canada, Burlington
Dodge, Doug	MNR, Fisheries Branch, Toronto
Dolan, Dave	IJC
Donahue, Michael	Great Lakes Commission
Dover, Peggy	WWF

Contact Name	Affiliation
Driver, Galen	OMAF
Eaton, Joan	Ontario Hydro
El-Shaarawi, A.	Environment Canada, Burlington
Elrod, Joe	US FWS, Cortland, NY
Estabrooks, Frank	Monroe County
Finley, Wally	Agriculture Canada
Finnie, Don	Finnie Distributing, St. Mary's, Ontario
Fleischer, Fred	MOE
Foley, Bob	NYDEC, Hale Creek Field Station
Fox, Glen	CWS, Ottawa
Frank, Dick	IJC
Gamble, Don	Rawson Academy of Aquatic Sciences, Ottawa
German, Murray	MOE, Bay of Quinte RAP coordinator
Gilbertson, Mike	IJC
Gilman, Andy	Health and Welfare, Ottawa
Grima, Lino	University of Toronto
Hallett, Doug	Ecologic
Hamilton, Andy	Rawson Academy of Aquatic Sciences, Ottawa, IJC
Hebert, Paul	Guelph
Henley, Doreen	Canadian Manufacturing Association
Hickey, John	USFWS, Cortland, NY
Jabonoski, Jean	EPO, City of Toronto
Jackson, John	Great Lakes United
Johannsson, Ora	DFO, Burlington
Johnson, Al	MOE
Kaiser, Klaus	Environment Canada, Burlington
Khan, Ansar	Public Works Canada, North York
Klose, Steve	MOE, Toronto RAP coordinator
Kramer, Jim	McMaster University
Krantzberg, Gail	MOE
Lefeveure, Al	Environment Canada, Burlington
LeTendre, Gerry	NYDEC, Lake Ontario Unit, Cape Vincent, NY
Litton, Simon	NYDEC
Mac, Michael	Great Lakes Environmental Research Lab., Ann Arbor
Mackay, Don	University of Toronto
Mann, Dianne	US Fish and Wildlife
Marr, Peter	MOE
Matzomotto, Mark	SUNY Buffalo
Maus, Laurie	Health and Welfare Canada, Ottawa

Contact Name	Affiliation
Mikol, Gerry	NYDEC
Miller, Sarah	Canadian Environmental Law Association
Mudroch, Alena	Environment Canada, Burlington
Muldoon, Paul	Pollution Probe
Munawar, Mohindra	Environment Canada, Burlington
Nalewajko, Czesia	University of Toronto
Neilson, Melanie	Environment Canada, Water Quality Branch, Burlington
Norstrom, Ross	CWS, Ottawa
Nugent, James	Monroe County Water Authority, Rochester
Palmer, Craig	US EPA
Palter, Jay	Greenpeace, Toronto
Persaud, Deo	MOE
Reynoldson, Trevor	Environment Canada, Burlington
Ricotta, Frank	Regional Water Engineer Rochester
Roberts, Ken	MOE
Rodgers, Keith	MOE, Hamilton RAP
Ryan, Tracy	Grand River Conservation Authority
Sanderson, Marie	University of Waterloo
Seto, Peter	MOE
Shimizu, Ron	University of Toronto and Environment Canada
Skinner, Larry	NYDEC, Bureau of Environmental Protection, Albany
Sly, Peter	Rawson Academy of Aquatic Sciences, Ottawa
Smardon, Richard	SUNY ESF, Syracuse
Smith, Janice	Environment Canada, Burlington
Smith, Ian	MOE
Smith, Lesbia	Ministry of Health
Somers, Keith	MOE, Great Lakes Section, Dorset, Ontario
Spencer, Vernon	OMAF
Sprules, Gary	University of Toronto, Erindale College
Stark, Alice	NYS Dept. of Health
Steer, Jim	Molson's
Stewart, Tom	MNR, Fisheries Branch, Glenora, Ontario
Stone, Ward	NYDEC, Wildlife Resources, Delmar Center
Suns, Karl	MOE
Swain, Wayland	Ecologic
Tapscott, Brian	OMAF, Guelph
Teet, Margaret	Monroe County, Dept. of Planning
Thomas, Richard	University of Windsor
Traper, Dick	NYDEC

Contact Name	Affiliation
Vallentyne, Jack	Environment Canada, Burlington
Villard, Serge	MOE
Villeneuve, David	Health and Welfare Canada, Ottawa
Wall, Greg	University of Guelph
Weaver, Grace	Kodak, Council of Great Lakes Industries
Weller, Phil	Great Lakes United, Buffalo
Wells, Peter	University of Toronto
Wesloh, Chip	CWS, Burlington, Ontario
Weston, Sandra	EPS, Environment Canada, Port Hope RAP coordinator
Willford, Wayne	US EPA
Wilson, K.	MOE
Wren, Chris	Ecological Planning Services, Guelph
Zafonte, Charles	US EPA
Zarull, Mike	Environment Canada

APPENDIX B

IMPAIRMENT OF BENEFICIAL USES IN LAKE ONTARIO

McMaster University
June 11, 1991

Workshop Participation List

A. Participants

Jannette Anderson
Environment Canada
Great Lakes Environment Office
25 St. Clair Avenue East, 6th Floor
Toronto, Ontario
M4T 1M2
(416) 972-

Victor Cairns
Manager, Fish Habitat Studies Division
Fisheries and Oceans, Ontario Region
867 Lakeshore Road, P.O. Box 5050
Burlington, Ontario
L7R 4A6
(416) 336-4862

Douglas Bryant
Assistant Professor
Dept. of Biochemistry
McMaster University
1200 Main Street West
Hamilton, Ontario
L8N 3Z5
(416) 525-9140, Ext. 2539

Murray N. Charlton
Project Chief, Lakes Restoration
Lakes Research Branch
National Water Research Institute
Canada Centre for Inland Waters
867 Lakeshore Road, P.O. Box 5050
Burlington, Ontario
L7R 4A6
(416) 336-4758

Dieter Busch
Coordinator, Lower Great Lakes
Fish and Wildlife Service, NY
Fisheries and Federal Aid
3075 Gracie Road
Cortland, NY 13045
(607) 753-1460

Thomas Coape-Arnold
RAP Coordinator
Ontario Ministry of the Environment
1 St. Clair Avenue West
Toronto, Ontario
M4V 1P5

Joseph de Pinto
Great Lakes Program
SUNY at Buffalo
207 Jarvis Hall
Buffalo, NY 14260
(716) 636-2088

Miriam Diamond
Research Associate
Institute for Environmental Studies
University of Toronto
Toronto, Ontario
M5S 1A4
(416) 978-6792

Michael Gilbertson
Physical Sciences Officer
International Joint Commission
100 Ouellette Avenue, Suite 800
Windsor, Ontario
N9A 6P3
(519) 256-7821

Andrew Hamilton
Director, Special Projects
Rawson Academy
1 Nicholas Street, Suite 404
Ottawa, Ontario
K1N 7B7
(613) 563-2636

John Holmes
Researcher
Institute for Environmental Studies
University of Toronto
Toronto, Ontario
M5S 1A4
(416) 978-2997

Klaus Kaiser
Environment Canada
Canada Centre for Inland Waters
867 Lakeshore Road, P.O. Box 5050
Burlington, Ontario
L7R 4A6
(416) 336-4756

Bruce Kershner
Great Lakes United
State University College at Buffalo
1300 Elmwood Avenue, Cassety Hall
Buffalo, NY 14222
(716) 886-0142

Nathalie La Violette
Researcher and Masters Student
Institute for Environmental Studies
University of Toronto
Toronto, Ontario
M5S 1A4
(416) 978-7898

Burkhard Mausberg
Pollution Probe
12 Madison Avenue
Toronto, Ontario
M5R 2S1
(416) 926-1907

Gerry Mikol
NY Department of Environment
Conservation
Great Lakes Section
50 Wolf Road
Albany, NY 12233-3502
(518) 457-0669

Melanie Neilson
Great Lakes Water Quality Division
Canada Centre for Inland Waters
867 Lakeshore Road, P.O. Box 5050
Burlington, Ontario
L7R 4A6
(416) 336-4963

Evert Nieboer
Professor of Toxicology
McMaster University
Dept. of Biochemistry
Health Sciences Centre
1200 Main Street West
Hamilton, Ontario
L8N 3Z5
(416) 525-9140, Ext. 2048

Lynn Norman
Environment Canada
Great Lakes Environment Office
25 St. Clair Avenue East, 6th Floor
Toronto, Ontario
M4T 1M2
(416) 972-4204

Darrell Piekarz
Environment Canada
Great Lakes Environment Office
25 St. Clair Avenue East, 6th Floor
Toronto, Ontario
M4T 1M2
(416) 972-

Henry Regier
Director
Institute for Environmental Studies
University of Toronto
Toronto, Ontario
M5S 1A4
(416) 978-4596

Laurie Sanderson
US Environmental Protection Agency
Region II
9311 Groh Road
Grosse Ille, MI 48138
(212) 264-5352

Griff Sherbin
Senior Advisor
Environment Canada
Great Lakes Environment Office
25 St. Clair Avenue East
Toronto, Ontario
M4T 1M2
(416) 973-1107

Ron Shimizu
Associate
Institute for Environmental Studies
University of Toronto &
Environment Canada
Toronto, Ontario
M5S 1A4
(416) 978-6409

Frances Silverman
Interim Director
Institute of Environment and Health
University of Toronto
Toronto, Ontario
M5S 1A4
(416) 978-5884

Sandy Slota
Researcher
Occupational Health
McMaster University
c/o 76 Maple Avenue, Apt. #8
St. Catharines, Ontario
L2R 2B2
(416) 684-5287

Ian Smith
Biohazards Scientist
Ontario Ministry of the Environment
125 Resources Road
Rexdale, Ontario
M9W 5L1
(416) 235-5789

Grace Wever
Eastman Kodak Company
RSPH - General Building 26
Floor 2
Rochester, NY 14652-3210
(716) 722-3348

Chip Weseloh
Canadian Wildlife Services
Canada Centre for Inland Waters
867 Lakeshore Road, P.O. Box 5050
Burlington, Ontario
L7R 4A6
(416) 336-4968

Don Williams
Inland Waters Directorate
Environment Canada
867 Lakeshore Road, P.O. Box 5050
Burlington, Ontario
L7R 4A6
(416) 336-4534

Charles Zafonte
US Environmental Protection Agency
Region II
9311 Groh Road
Grosse Ille, MI 48138
(212) 264-5352

Michael Zarull
Great Lakes Officer
Lakes Research Branch
Canada Centre for Inland Waters
867 Lakeshore Road, P.O. Box 5050
Burlington, Ontario
L7R 4A6
(416) 336-4783

B. Reviewers of Workshop Material

Laurie Maus
Great Lakes Health Effects Program
Health and Welfare Canada
Environmental Health Centre
Tunney's Pasture
Ottawa, Ontario
K1A 0L2
(613) 957-9154

APPENDIX C
IMPAIRMENT OF BENEFICIAL USES IN LAKE ONTARIO
TUESDAY JUNE 11, 1991 – McMASTER UNIVERSITY
WORKSHOP AGENDA
ROOM 3E26

8:30 a.m. *COFFEE AND INTRODUCTIONS*

9:00 WELCOME (Henry Regier and Frances Silverman)

Introduction to the Institute of Environment and Health
Introduction to the study: goals, approach
Description of the goals of the workshop

9:30 CRITICAL APPRAISAL GUIDELINES: PRESENTATION AND DISCUSSION (Evert Nieboer and Doug Bryant)

History, need for critical appraisal guidelines
Development of project guidelines

10:25 SUMMARY OF WORKSHOP INPUT ON GUIDELINES (Mike Gilbertson)

10:35 *COFFEE BREAK*

10:55 INVENTORY OF IMPAIRMENT OF 3 BENEFICIAL USES: PRESENTATION AND DISCUSSION (Sarah Rang, Sandy Slota and John Holmes)

Impairment #5: Bird and animal deformities and reproductive problems
Impairment #4: Fish tumors and other deformities
Impairment #10: Beach closures

12:00 *LUNCH*

1:30 SUMMARY OF WORKSHOP INPUT ON INVENTORY (Mike Gilbertson)

1:45 PRACTICAL IMPLICATIONS AND SYNTHESIS: PRESENTATION AND DISCUSSION (Evert Nieboer)

3:00 SUMMARY OF DISCUSSIONS AT THE WORKSHOP (Henry Regier)

3:15 FORMAL MEETING ADJOURNS, COFFEE BREAK

3:30 INFORMAL DISCUSSIONS (all are welcome)

APPENDIX D

LIST OF COMMON AND SCIENTIFIC NAMES USED IN THE STUDY

Common Name	Scientific name
Fish	
coho salmon	*Oncorhynchus tshawytscha*
chinnook salmon	*Oncorhynchus kisutch*
rainbow (steelhead) trout	*Oncorhynchus mykiss*
lake trout	*Salvelinus namaycush*
brown trout	*Salmo trutta*
Atlantic salmon	*Salmo salar*
lake whitefish	*Coregonus clupeaformis*
ciscoes	*Coregonus spp.*
burbot	*Lota lota*
slimy sculpin	*Cottus cognatus*
rainbow smelt	*Osmerus mordax*
sea lamprey	*Petromyzon marinus*
gizzard shad	*Dorosoma cepedianum*
alewife	*Alosa pseudoharengus*
American eel	*Anguilla rostrata*
white sucker	*Catostomus commersoni*
white perch	*Morone americana*
white bass	*Morone chrysops*
freshwater drum	*Aplodinotus grunniens*
channel catfish	*Ictalurus punctatus*
brown bullhead	*Ictalurus nebulosus*
walleye	*Stizostedion vitreum*
yellow perch	*Perca flavescens*
northern pike	*Esox lucius*
smallmouth bass	*Micropterus dolomieui*
largemouth bass	*Micropterus salmoides*
carp	*Cyprinus carpio*
goldfish	*Carassius auratus*
carp x goldfish	
spottail shiner	*Notropis hudsonius*
emerald shiner	*Notropis atherinoides*

APPENDIX D. Continued.

Common Name	Scientific name
Colonial Fish-eating Birds	
double-crested cormorant	*Phalacrocorax auritus*
black-crowned night-heron	*Nycticorax nycticorax*
bald eagle	*Haliaeetus leucocephalus*
herring gull	*Larus argentatus*
ring-billed gull	*Larus delawarensis*
Caspian tern	*Sterna caspia*
common tern	*Sterna hirundo*
Mammals	
mink	*Mustela vison*
Reptiles	
snapping turtle	*Chelydra serpentina*

Appendix E

SUMMARY OF STUDIES ON FISH TUMORS OR OTHER DEFORMITIES IN LAKE ONTARIO

Reference	Sonstegard (1977)	Black (1988)
Location	Western Lake Ontario	Lake Ontario (NS)
Time of Collection	1973-1976	<1984
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	Incidence of lip papillomas in spawning fish: Hamilton-35.1%; Dufferin Ck-0%; Toronto-2.2%; Oakville Ck-50.8%; Jordan Harbour-8.0%; Rochester 0.8%. Viral etiology but chemicals involved.	Incidence and size of epidermal and/or oral papillomas increases in urbanized western basin and position of lesion shifts onto or about the lips in heavily polluted areas.
Contaminant	NI	Organochlorines suspected
Contaminant source	NI	NI
Type of Report	Secondary	Secondary
Type of Evidence	Observational	n/a

Reference	Government of Canada (1991b)	Smith et al. (1989a)
Location	Spencer Creek (NS)	Grindstone Creek (NS)
Time of Collection	1983-87	1985
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	63.8% of spawning fish 7 years or older had lip papillomas (N=191).	Incidence of epidermal papillomas was 58% in 225 spring spawning run fish. 43% had lip papillomas, 20% plaques, and 16% body papillomas.
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Appendix E. Continued.

Reference	Government of Canada (1991b)	Smith et al. (1989a)
Location	Grindstone Creek (NS)	Oakville Creek (NS)
Time of Collection	1983-87	1985-87
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	39.0% of spawning fish 7 years or older had lip papillomas (N=1200).	Incidence of epidermal papillomas was 59% in 482 spring spawning fish. 46% had lip papillomas, 21% plaques, & 13% body papillomas. Prevalence increased in fish > 3-4 yrs.
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Reference	Smith et al. (1989b)	Government of Canada (1991b)
Location	Oakville Creek (NS)	Humber River (NS)
Time of Collection	ca. 1988	1983-87
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	Epidermal papillomas regressed 22-79% depending on type. New tumours developed in lab, may mean non-contaminant etiology (viral?).	24.0% of spawning fish 7 years or older had lip papillomas (N=317).
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Secondary	Primary
Type of Evidence	Experimental	Observational

Appendix E. Continued.

Reference	Government of Canada (1991b)	Government of Canada (1991b)
Location	Cobourg Creek (NS)	Trent River at Bay of Quinte (NS)
Time of Collection	1983-87	1983-87
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	14.4% of spawning fish 7 years or older had lip papillomas (N=152).	4.4% of spawning fish 7 years or older had lip papillomas (N=183).
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Reference	Smith and Zajdlik (1987)	Smith and Ferguson (1985)
Location	Oakville Creek (NS)	Grindstone Creek (NS)
Time of Collection	1986	1985
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	Epidermal papillomas regressed 22-79% in lab depending on type. New tumours developed on 23 and 60% of fish in two groups.	Fish had higher rates of external (23.6%), lip (24.4%), and internal (4.1%) neoplasms than fish from Whites Creek, a reference site from Lake Simcoe.
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Primary	Secondary
Type of Evidence	Observational	Observational

Appendix E. Continued.

Reference	Hayes et al. (1990)	Government of Canada (1991b)
Location	Grindstone/Oakville Creeks pooled (NS)	Forty-Mile Creek (NS)
Time of Collection	NI	1984
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	Incidence of epidermal papillomas (N=707) was about 60%, incidence of bile duct tumours (N=456) was about 5%, & hepatocellular tumours (N=456) was 1%. Biliary disease may promote development.	0% incidence of total liver tumours in 133 fish in the spawning run.
Contaminant	NI - not chemicals detox by GST	NI
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational & Experimental	Observational

Reference	Government of Canada (1991b)	Government of Canada (1991b)
Location	Grindstone Creek (NS)	Oakville Creek (NS)
Time of Collection	1982, 1984, 1985	1984, 1985
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	Incidence of total liver tumours was 1.2% in 168 spawning fish. 1.7% cholangiolar carcinomas, 0.9% hepatocellular tumours, & 0.9% adenomas.	Incidence of total liver tumours was 7.4% in 612 spawning fish. 6.5% cholangiolar carcinomas, 2.8% hepatocellular tumours, & 2.2% adenomas.
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Appendix E. Continued.

Reference	Government of Canada (1991b)	Government of Canada (1991b)
Location	Humber River (NS)	Rouge River (NS)
Time of Collection	1987	1987
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	Incidence of total liver tumours was 4.7% in 192 fish collected from the spring spawning run.	Incidence of total liver tumours was 3.5% in 199 fish collected from the spring spawning run.
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Reference	Government of Canada (1991b)	Government of Canada (1991b)
Location	Cobourg Creek/Ganaraska River pooled (NS)	Trent River (NS)
Time of Collection	1987	1987
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	Incidence of total liver tumours was 6.0% in 116 fish collected from the spring spawning run.	Prevalence of total liver tumours was 0.7% in 148 fish collected from the spring spawning run.
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Appendix E. Continued.

Reference	Kirby et al. (1988)	Kirby et al. (1989b)
Location	Oakville Creek (NS)	Oakville Creek (NS)
Time of Collection	ca. 1987	ca. 1988
Species	*Catostomus commersoni*	*Catostomus commersoni*
Findings	B(a)P excreted 2X as fast as reference fish within 7 days. GST activity lower than in reference fish. May relate to liver tumour induction and proliferation.	Bile duct and hepatocellular adenomas or carcinomas deficient in GST proteins. This may be related to PAH exposure and subsequent tumour growth.
Contaminant	B(a)P suspected	Benzo(a)pyrene
Contaminant source	NI	Sediments
Type of Report	Secondary	Secondary
Type of Evidence	Experimental	Experimental

Reference	Kirby et al. (1989a)	Sonstegard (1977)
Location	Oakville Creek and Humber River (NS)	Lake Ontario
Time of Collection	1988	1973-1976
Species	*Catostomus commersoni*	carp x goldfish hybrids
Findings	Higher levels of B(a)P and metabolites relative to fish from reference sites. GST is a major detoxification pathway. Relates to development of liver tumours.	Hybrids exhibit epizootics of gonadal tumours with incidence approaching 100% in older males. Tumour size and prevalence increase with age after 3-4 years old.
Contaminant	Benzo(a)pyrene	NI
Contaminant source	Probably sediments	NI
Type of Report	Secondary	Secondary
Type of Evidence	Experimental	Observational

Appendix E. Continued.

Reference	Dickman and Steele (1986)	Down et al. (1988)
Location	Welland River	Cootes Paradise and Hamilton Harbour (NS)
Time of Collection	1977-1978	1978-1981
Species	carp x goldfish hybrids	carp x goldfish hybrids
Findings	General correlation between industrial loadings & prevalence of gonadal neoplasms. Section with highest loadings had 48% incidence, 80% in sexually mature hybrids.	Hybrids have epizootic gonadal tumours, which are rare in parental species. Non-specific nature suggests factors other than chemicals involved in initiation and development.
Contaminant	NI	NI
Contaminant source	Probably heavy industries	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Experimental

Reference	Smith and Ferguson (1985)	Smith et al. (1989a)
Location	Hamilton Harbour (NS)	Hamilton Harbour
Time of Collection	1985	1985
Species	carp x goldfish hybrids	*Ictalurus nebulosus*
Findings	Gonadal tumours present in 4 of 5 fish versus 18 of 35 from Long Point reference site.	Incidence of epidermal papillomas was 55% in 176 fish in summer, compared to 15% of 53 fish from Long Point, an unpolluted site.
Contaminant	NI	PAH suspected
Contaminant source	NI	NI
Type of Report	Secondary	Primary
Type of Evidence	Observational	Observational

Appendix E. Continued.

Reference	Black (1983)	Smith and Ferguson (1985)
Location	Buffalo River	Hamilton Harbour
Time of Collection	1980-81	1985
Species	*Ictalurus nebulosus*	*Ictalurus nebulosus*
Findings	At 18 months 5 of 22 fish were hyperplastic and 2 of 5 had skin papillomas over area exposed to sediment extract. This plus sediment PAH and neoplasms in feral fish are correlated.	External growths seen on 48% of fish and liver lesions observed in 7%. Fish from Long Point reference site had rates of 14% and 0% respectively.
Contaminant	PAHs	NI
Contaminant source	Buffalo River sediments	NI
Type of Report	Primary	Secondary
Type of Evidence	Observational and Experimental	Observational

Reference	Hayes et al. (1990)	Government of Canada (1991b)
Location	Hamilton Harbour (NS)	Cayuga Creek (NS) (Niagara River)
Time of Collection	NI	1982-87
Species	*Ictalurus nebulosus*	*Ictalurus nebulosus*
Findings	Incidence of epidermal papillomas (N=176) was about 55%, bile duct tumours (N=170) was about 5%, & hepatocellular tumours (N=170) about 0.5%.	20.0% incidence of liver tumours in 40 fish.
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Appendix E. Continued.

Reference	Government of Canada (1991b)	Government of Canada (1991b)
Location	Buffalo River (NS) (Niagara River)	Black Creek (NS) (Niagara River)
Time of Collection	1982-87	1982-87
Species	*Ictalurus nebulosus*	*Ictalurus nebulosus*
Findings	16.7% incidence of liver tumours in 36 fish.	<1.0% incidence of liver tumours in 101 fish.
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Reference	Government of Canada (1991b)	Maccubbin et al. (1988)
Location	Hamilton Harbour (NS)	Niagara River (NS)
Time of Collection	1982-87	ca. 1987
Species	*Ictalurus nebulosus*	*Ictalurus nebulosus*
Findings	2.0% incidence of liver tumours in 124 fish.	Metabolites of PAH (BaP &PHE) were detectable in bile within 24 hrs of feeding. BaP & PHE levels 7-148 X and 7-297 x > than controls. PAH exposure may account for high neoplasia.
Contaminant	NI	B(a)P & PHE
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Experimental & Observational

Appendix E. Continued.

Reference	Moccia et al. (1977)	Metcalfe et al. (1990)
Location	Lake Ontario (Credit R.?)	Hamilton Harbour (NS)
Time of Collection	1976	NI
Species	*Oncorhynchus kisutch*	*Oncorhynchus mykiss*
Findings	Fish collected from fall spawning run (N=63) had overt goiter frequency of 46%, up from 24% (N=51) in fish from 1975 run. Iodine deficiency is a factor & also pollutants.	Induced hepatocellular tumours by injecting sac fry with sediment extract. Incidence was dose-dependent, 3.1-8.1% (N=65-148). No tumours formed in fish given Oakville Ck sediment extract. PAH-type chemicals suspected
Contaminant	NI	NI
Contaminant source	NI	Primary
Type of Report	Primary	Experimental
Type of Evidence	Observational	

Reference	Metcalfe et al. (1988)	Ruby and Cairns (1983)
Location	Hamilton Harbour (NS)	Yorkshire Bar (NS in eastern Lake Ontario)
Time of Collection	NI	1979
Species	*Oncorhynchus mykiss*	*Salvelinus namaycush*
Findings	Sediment extracts induced liver tumours in 3.1 to 8.1% of fish, depending on the dose. Sediment extracts were mutagenic and carcinogenic.	5 of 8 male fish collected during fall spawning had 1 or more partial or complete testicular constrictions. Prevelance in western end was 38% (N=167 males).
Contaminant	PAHs suspected	NI - chemicals one possibility
Contaminant source	sediment extracts	NI
Type of Report	Primary	Primary
Type of Evidence	Experimental	Observational

Appendix E. Continued.

Reference	Black (1984)
Location	Great Lakes (NS)
Time of Collection	1970s & 1980s
Species	Various fish species
Findings	Neoplasms observed in 8 species in heavily polluted areas of the Great Lakes. Probable chemical etiology but only direct evidence relates to brown bullheads in Buffalo R.
Contaminant	PAH - brown bullhead
Contaminant source	Steel, coking, dye mfgs.
Type of Report	Secondary
Type of Evidence	n/a

Species mentioned:
C. commersoni - white sucker
I. nebulosus - brown bullhead
O. kisutch - coho salmon
O. mykiss - rainbow trout
S. namaycush - lake trout

General Codes:
NI - not indicated in paper
NS - nearshore location
OS - offshore location

Appendix F

SUMMARY OF STUDIES ON BIRD OR ANIMAL DEFORMITIES OR REPRODUCTIVE PROBLEMS IN LAKE ONTARIO

Access Code	Gilbertson (1974)	Teeple (1977)
Location	6 colonies in Great Lakes	Brothers Is. (Eastern L.Ont)
Time of Collection	1972	1973
Species	*Larus argentatus*	*Larus argentatus*
Findings	reproductive problems rate = 0.06 -0.21 chick/pair in LO vs approx 1 in control 9-16 % eggshell thinning, correlated with DDE flaking, breakage of eggs	reproductive problems 77% of eggs failed to hatch low reproductive success (0.06 -0.018 chicks/pair)
Contaminant	DDE and PCB	DDE,PCB, HCB,Hg etc.
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Access Code	Gilbertson (1982)	Gilbertson and Hale (1974a)
Location	Scotch Bonnet Is.	Scotch Bonnet Is.
Time of Collection	1973, 1974	1973
Species	*Larus argentatus*	*Larus argentatus*
Findings	reproductive problems embryo abnormalities, chick edema in 36% of GI vs 8% in control, cloudy fluid in peritoneal cavity, shorter tarsi, increased liver wt, porphyrin	embryo abnormalities 22% mortality vs 3% in U.S., higher embryo mortality in Great Lakes than other regions
Contaminant	TCDD	NI
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Appendix F. Continued.

Access Code	Gilbertson and Hale (1974b)	Gilman et al. (1977)
Location	Scotch Bonnet Is.	Scotch Bonnet Is.
Time of Collection	1973	1974 -1975
Species	*Larus argentatus*	*Larus argentatus*
Findings	reproductive problems 16% hatchability	reproductive problems reproductive rate = 1/10 of other Great Lakes colonies more eggs disappear more embryos fail to hatch incubation shows intrinsic factor
Contaminant	NI	PCB, mirex, DDE
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Access Code	Gilbertson and Fox (1977)	Moccia et al. (1986)
Location	Lake Ontario	Gull Is., Scotch Bonnet Is. Presqu'ile
Time of Collection	1974	1974 -SB, 1980 -1983 GI
Species	*Larus argentatus*	*Larus argentatus*
Findings	embryo abnormalitites, reduced in size, enlarged livers increased porphyria, peritoneal and pericaudal fluid, subcutaneous muco-serous exudates, tarsus length correlated with DDE	deformities, goiters - higher thyroid mass in Great Lakes vs control, smaller follicle diameter, taller epithelial cells, less colloid, higher prevalence of cell hyperplasmia
Contaminant	PCB, HCB, DDE, chlordane	PHAH (not PCB's)
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Appendix F. Continued.

Access Code	Gilbertson et al. (1976)	Government of Canada (1991b)
Location	Scotch Bonnet Is., Pigeon, Presqu'ile, Muggs	Lake Ontario
Time of Collection	1972 -1973	1970's
Species	*Larus argentatus*	*Larus argentatus*
Findings	chick abnormalities 1 /850 chicks PCB conc = 580 ppm	super normal clutches
Contaminant	PCB	NI
Contaminant source	NI	NI
Type of Report	Primary	Tertiary
Type of Evidence	Observational	Observational

Access Code	Government of Canada (1991b)	Peakall and Fox (1987)
Location	Lake Ontario	Great Lakes
Time of Collection	1975- 1976	1972 -1980
Species	*Larus argentatus*	*Larus argentatus*
Findings	feminisation 5/7 females were feminized 5/10 males had abnormally enlarged oviducts	reproductive problems reduced nest temperature reduced nest attentivenes egg exchange experiments showed clean adults unable to hatch dirty eggs, and vv
Contaminant	NI	Organochlorines
Contaminant source	NI	NI
Type of Report	Tertiary	Primary
Type of Evidence	Observational	Observational

Appendix F. Continued.

Access Code	Government of Canada (1991b)	Government of Canada (1991b)
Location	Lake Ontario	Lake Ontario
Time of Collection	1981	1980 -1984
Species	*Larus argentatus* embryos	*Larus argentatus*
Findings	biochemical alterations, elevated AHH activity in gull livers vs control	biochemical alterations reduced retinol and retinyl palmitate levels in GL vs control 1985 retinol levels higher than 1982 levels
Contaminant	NI	NI
Contaminant source	NI	NI
Type of Report	Tertiary	Tertiary
Type of Evidence	Observational	Observational

Access Code	Government of Canada (1991b)	Spear et al. (1986)
Location	Lake Ontario	Presqu'ile
Time of Collection	1980 -1984	1982
Species	*Larus argentatus*	*Larus argentatus*
Findings	biochemical alterations degree of thyroid enlargement and prevalence of thryroid hyperplasmia is correlated with PCB	biochemical alterations, Lake Ontario had lowest mean retinol conc. and highest TCDD conc., different than control, dioxin analogue reduced retinol conc. and increased AHH activity
Contaminant	PCB	2,3,7,8-TCDD
Contaminant source	NI	NI
Type of Report	Tertiary	Primary
Type of Evidence	Observational	Observational and Experimental

Appendix F. Continued.

Access Code	Fox et al. (1988)	Spear et al. (1990)
Location	Hamilton Harbour, Snake Is.	Eastern Lake Ontario Pigeon Is., Snake Is.
Time of Collection	1980 -1987	1987
Species	*Larus argentatus*	*Larus argentatus*
Findings	significantly elevated levels of HCP (highly carboxylated porphyrins) in livers of gulls from L.Ont vs control, HCP levels not correlated to 1 contaminant, chicks also had high HCP levels	biochemical abnormalities molar ratio of retinol to retinyl palmitate in yolk of eggs was correlated to indices of PCDD and PCDF
Contaminant	PAH	PCDD and PCDF
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Access Code	Government of Canada (1991b)	Gilbertson et al. (1976)
Location	Pigeon Island	Pigeon Island
Time of Collection	1961 -1973	1972 -1973
Species	*Nycticorax nycticorax*	*Nycticorax nycticorax*
Findings	reproductive problems 1969 total failure 1972 0.22 yg/pr eggshell thinning 14 -16 % from 1972 -1976 1982 8% thinner	chick abnormalities 1 / 39 chicks crossbill PCB conc = 304 ppm
Contaminant	DDE	PCB
Contaminant source	NI	NI
Type of Report	Tertiary	Primary
Type of Evidence	Observational	Observational

Appendix F. Continued.

Access Code	Gilbertson et al. (1976)	Government of Canada (1991b)
Location	Muggs Island Pigeon Is.	Southern Lake Ontario
Time of Collection	1972 -1973	1969, 1973
Species	*Larus delawarensis*	*Larus delawarensis*
Findings	chick abnormalities 24 chicks leg deformity PCB conc = 379 ppm 1/500 chicks bill deformity	Population dieoff
Contaminant	PCB	PCB, DDE, dieldrin
Contaminant source	NI	NI
Type of Report	Primary	Tertiary
Type of Evidence	Observational	Observational

Access Code	Gilbertson et al. (1976)	Government of Canada (1991b)
Location	Muggs Is., Presqu'ile, Scotch Bonnet, Brother, Pigeon Is.	Lake Ontario
Time of Collection	1972 -1973	1971 -1973
Species	*Sterna hirundo*	*Sterna hirundo*
Findings	chick abnormalities < 1% abnormalities various abnormalitites PCB conc = 155 ppm in clean, 257 ppm in colonies with abnorm.	birth defects 130/10000 chicks highest observed rate ever
Contaminant	PCB	NI
Contaminant source	NI	NI
Type of Report	Primary	Tertiary
Type of Evidence	Observational	Observational

Appendix F. Continued.

Access Code	Government of Canada (1991b)	Gilbertson et al. (1976)
Location	Lake Ontario	Pigeon Island
Time of Collection	1972, 1976, 1977	1972 -1973
Species	*Sterna hirundo*	*Sterna caspia*
Findings	reproductive problems 1972 0.09 -0.13 yg/pr 1976 0.55 -0.92 yg/pr 1977 1.18 yg /pr	chick abnormalities 1/100 chicks crossbill PCB conc = 359 ppm
Contaminant	NI	PCB
Contaminant source	NI	NI
Type of Report	Tertiary	Primary
Type of Evidence	Observational	Observational

Access Code	Weekes (1975)	Government of Canada (1991b)
Location	Eastern Lake Ontario and Lake Erie	Delta, Lake Ontario
Time of Collection	1974	1969, 1973
Species	*Haliaeetus leucocephalus*	*Haliaeetus leucocephalus*
Findings	reproductive problems LO nest did not produce young LE rate = 50% in 1974 vs 20% from 1969 -1974	reproductive problems 1/12 nest produced young no nests along LO in 1980's
Contaminant	DDT	
Contaminant source	NI	NI
Type of Report	Primary	Tertiary
Type of Evidence	Observational	Observational

Appendix F. Continued.

Access Code	Gilbertson (1989b)	Price and Weseloh (1986)
Location	Lake Ontario	Eastern Lake Ontario
Time of Collection	future	1978 -1982
Species	*Haliaeetus leucocephalus*	*Phalacrocorax auritus*
Findings	ecosystem objective 20 pairs on LO, with rate = 1 chick/pair, contaminant burdens for eggs and adults given Techn. chlordane	reproductive problems DDE levels decreased 50% eggshell thinning decreased from 24 to 6% rate = 1.4 -1.4
Contaminant	DDT, PCB, heptachlor epoxide,	
Contaminant source	NI	NI
Type of Report	Tertiary	Primary
Type of Evidence	Policy for field	Observational

Access Code	Government of Canada (1991b)	Leatherland and Sonstegard (1980)
Location	Lake Ontario and Great Lakes	Lake Ontario, Lake Erie
Time of Collection	1979 -1987	NI
Species	*Phalacrocorax auritus*	Sprague-Dawley rats
Findings	chick deformities cross bill deformity 3.5/10000 chicks vs 2.4 and 0.6 in controls	biochemical changes rats fed coho had enlarged thyroid, larger epithelial cells, highly vacuolated cells hypertrophy and hyperplasia of thyroid
Contaminant	PCB	PCB
Contaminant source	NI	NI
Type of Report	Tertiary	Primary
Type of Evidence	Observational	Experimental

Appendix F. Continued.

Access Code	Government of Canada (1991b)	Aulerich and Ringer (1977)
Location	Lake Ontario, Lake Erie	lab
Time of Collection	1986 -1989	1968 -1974
Species	*Chelydra serpentina*	*Mustela vision*
Findings	birth defects, higher unhatched and deformed young vs control, unhatched eggs range 10 -45% vs <20% in control, deformed range 5-40% vs 5% in control	reproductive problems 5, 10 , 15 ppm PCB caused reproductive failure 2 ppm reduced reproduction reproductive failure is not permanent
Contaminant	NI	PCB
Contaminant source	NI	NI
Type of Report	Tertiary	Primary
Type of Evidence	Field	Experimental

Access Code	Foley et al. (1988)	Species mentioned:
Location	Eastern Lake Ontario and United States	*L. argentatus* - herring gull *L. delawarensis* - ring-billed gull
Time of Collection	1982 - 1984	*N. nycticorax* - black-crowned night-heron
Species	*Mustela vision*	*P. auritus* - double-crested cormornat
Findings	possible reproductive failure PCB levels in mink approach lab toxic dose Hg not implicated	*S. hirundo* - common tern *S. caspia* - Caspian tern *H. leucocephalus* - bald eagle *M. vison* - mink
Contaminant	PCB	*C. serpentina* - snapping turtle
Contaminant source	NI	
Type of Report	Primary	General Codes:
Type of Evidence	Observational	NI - not indicated in paper NS - nearshore location OS - offshore location

APPENDIX G

SUMMARY OF DREDGING PROJECTS IN LAKE ONTARIO WATERS, 1975-1989

Location	Kingston (Treasure Island Marina)	Long Point (Amherst Is.)
Date	1989	1978
Equipment	mechanical	clam
Disposal	upland	open lake
Material	sand, silt	gravel, sand
Quantity (m3)	25,075	1,800
Chemical data	VS-35.9%; OG-0.13; TP-0.75; TKN-1.1%; COD-53.9%; PCB-<0.01; Hg-0.055; Pb-18.6; As-2.8; Cd-0.56; Cu-34.8; Zn-133.9; Cr-95.1; Ni-37.06	Maintenance dredging - no chemical data collected
Reference	Sediment Work Group (1991)	Dredging Subcommittee (1982)

Location	Trent-Severn Waterway	Trenton Bay (marina)
Date	1980	1989
Equipment	hydraulic	mechanical backhoe
Disposal	confined	upland
Material	silt, clay	sand, silt
Quantity (m3)	23,721	5,093
Chemical data	OG-3.1; VS-41.0%; TP-0.66; TKN-0.23; COD-228.0; Hg-0.124; Samples collected in 1979	VS-4.2%; OG-0.06; TP-0.82: TKN-1.4; PCB-<0.01; Hg-0.047; Pb-28.9; As-4.4; Cd-0.88; Cu-13.4; Zn-18.8; Cr-34.7; Ni-18.8
Reference	Sediment Work Group (1990)	Sediment Work Group (1991)

Appendix G. Continued.

Location	Point Traverse	Point Traverse
Date	1976-77	1982
Equipment	clam, dragline	clam and scow
Disposal	open lake & land (~1,000)	open lake
Material	sands, gravel	Class B
Quantity (m3)	4,800	2,417
Chemical data	No chemical data	OG-0.98; TP-0.42; TKN-1.72; COD-41.03; PCB-0.088; Cd-1.4; Cu-9.0; Zn-30.6; Cr-12.2
Reference	Dredging Subcommittee (1982)	Sediment Work Group (1990)

Location	Point Traverse	Cobourg (approach)
Date	1988	1976
Equipment	mechanical clamshell	clam
Disposal	upland	open lake
Material	sand, gravel	sand, silt, clay
Quantity (m3)	1,303	33,592
Chemical data	VS-2.4%; OG-0.077; TP-0.43; COD-1.7%; TKN-1.1; PCB-0.03; Hg-0.06; Pb-16.0; As-1.4; Cd-0.49; Cu-7.6; Zn-23.8; Cr-21.8; Ni-12.1	VS-2.3%; OG-0.52; TP-0.70 COD-20.8; TKN-0.60; Hg-0.12
Reference	Sediment Work Group (1991)	Dredging Subcommittee (1982)

Appendix G. Continued.

Location	Cobourg (approach)	Cobourg (approach)
Date	1977	1978
Equipment	clam	dipper
Disposal	open lake	open lake
Material	sand, silt-clay	sand, silt
Quantity (m3)	16,000	14,400
Chemical data	VS-2.3%; OG-0.52; TP-0.7; COD-20.8; TKN-0.60; Hg-0.12	VS-2.3%; OG-0.52; TP-0.70; COD-20.8; TKN-0.60; Hg-0.12
Reference	Dredging Subcommittee (1982)	Dredging Subcommittee (1982)

Location	Port Hope	Oshawa (cruise marina)
Date	1979	1978
Equipment	clam	backhoe
Disposal	open lake	land, by truck in harbour
Material	sand, silt	organic clay & debris
Quantity (m3)	28,000	65,000
Chemical data	VS-2.7%; OG-0.46; COD-19.0; TP-0.63	No chemical data collected
Reference	Dredging Subcommittee (1982)	Dredging Subcommittee (1982)

Appendix G. Continued.

Location	Oshawa (approach)	Oshawa (approach)
Date	1978	1977
Equipment	clam	hopper
Disposal	open lake	open lake
Material	sand, silt	sand, silt
Quantity (m3)	34,500	10,200
Chemical data	VS-2.5%; OG-0.693; TP-0.59; COD-24.5; TKN-0.58; Hg-0.06	Same as in 1978
Reference	Dredging Subcommittee (1982)	Dredging Subcommittee (1982)

Location	Oshawa	Oshawa
Date	1979	1980
Equipment	hydraulic	scows
Disposal	confined east of wharf	open lake
Material	silt, clay	gravel, sand
Quantity (m3)	61,000	12,647
Chemical data	VS-5.9%; OG-1.56; TP-1.06; COD-80.8; TKN-1.59 Hg-0.13; Pb-55.0; As-2.8; Cu-14.6; Zn-124.0; Cr-129.0; Ni-79.0; PCB-1.267	OG-0.4; VS-2.2%; TP-0.38; TKN-0.48; COD-17.4; PCB-0.033; Hg-0.023; Pb-39.2; Cu-21.8; Zn-55.6; Cr-112.4; Ni-42.5
Reference	Dredging Subcommittee (1982)	Sediment Work Group (1990)

Appendix G. Continued.

Location	Oshawa	Oshawa
Date	1982	1983
Equipment	hydraulic	clam and scow
Disposal	confined	confined
Material	sand, silt, clay	sand, silt
Quantity (m3)	13,866	7,714
Chemical data	OG-1.29; TP-0.44; TKN-0.62; COD-24.5; PCB-0.26; Hg-0.028; Pb-39.8; Cd-1.0; Cu-12.2; Zn-49.2; Cr-29.2; Ni-34.1	OG-0.99; VS-2.4%; TP-0.68; TKN-1.3; COD-22.0; PCB-0.07; Pb-32.0; Cd-4.6; Cu-9.6; Zn-55.0; Cr-27.3; Ni-27.0
Reference	Sediment Work Group (1990)	Sediment Work Group (1990)

Location	Oshawa (inner harbour/east wharf)	Oshawa (approach & entrance channels)
Date	1985	1986
Equipment	scow	clamshell
Disposal	confined	open lake
Material	sand, silt	silt, sand, clay
Quantity (m3)	13,331	20,784
Chemical data	VS-4.95%; OG-0.89; TP-0.86; TKN-1.08; PCB-0.16; Hg-0.03; Pb-56.5; As-1.95; Cd-0.5; Cu-13.5; Zn-127.5; Cr-65.0; Ni-46.0 Samples collected in 1984	VS-1.8%; OG-0.32; TP-0.53; TKN-0.50; PCB-<0.05; Hg-0.023; Pb-14.1; As-2.1; Cd-<1.0; Cu-9.6; Zn-41.6; Cr-18.3; Ni-8.6
Reference	Sediment Work Group (1991)	Sediment Work Group (1991)

Appendix G. Continued.

Location	Oshawa (approach/outer channel)	Whitby
Date	1988	1978
Equipment	derrick/clamshell bagotuille	hydraulic
Disposal	open lake	confined in harbour
Material	sand	organic peat, clay, silt
Quantity (m3)	16,362	188,300
Chemical data	VS-2.07%; OG-0.6; TP-0.35; TKN-0.3; COD-1.1%; PCB-<0.01 Hg-<0.01; Pb-17.2; As-1.5; Cd-0.31; Cu-7.28; Zn-21.4; Cr-11.8; Ni-14.4	VS-18.2%; COD-271.0; OG-1.74; TKN-3.5; TP-0.86; PCB-0.18; Hg-0.15; Pb-73.0; As-4.5; Cu-83.0; Ni-82.0; Zn-207.0; Cr-155.0
Reference	Sediment Work Group (1991)	Dredging Subcommittee (1982)

Location	Whitby	Toronto Harbour
Date	1989	pre-1979
Equipment	suction	clam
Disposal	confined	confined disposal facility
Material	sand, silt, clay	silt, clay
Quantity (m3)	5,362	64,495
Chemical data	VS-8.3%; OG-0.69; TP-0.49; COD-14.4; TKN-0.40; PCB-<0.05; Hg-0.19; Pb-16.0; As-1.6; Cd-0.80; Cu-21.5; Zn-0.099; Cr-40.5; Ni-27.0	VS-7.1%; TKN-1.7; OG-4.2; TP-1.18; PCB-0.23; Pb-175.0; Zn-230.0; Ni-18.0; Hg-0.25; Cu-45.0; Cr-47.0; Based on samples from 1972, 1973, 1976, 1978
Reference	Sediment Work Group (1991)	Dredging Subcommittee (1982)

Appendix G. Continued.

Location	Toronto Harbour	Toronto Harbour
Date	1980	1981
Equipment	clam	clam
Disposal	confined	confined
Material	unknown	unknown
Quantity (m3)	43,045	102,875
Chemical data	No chemical data, relied on sampling from 1972, 1973, 1976, and 1978	No chemical data, relied on sampling from 1972, 1973, 1976, and 1978
Reference	Sediment Work Group (1990)	Sediment Work Group (1990)

Location	Toronto Harbour (east entrance)	Port Credit
Date	1983	1975-76
Equipment	dipper, scow	clam
Disposal	open lake	confined disposal
Material	Class B sand	organic silts
Quantity (m3)	31,428	4,700
Chemical data	OG-0.14; PCB-0.02; Hg-0.01; Pb-15.0; As-1.28; Cd-0.5	VS-4.2%; OG-3.3; TP-0.33 COD-52.0; TKN-1.8
Reference	Sediment Work Group (1990)	Dredging Subcommittee (1982)

Appendix G. Continued.

Location	Port Credit	Port Credit (approach channel)
Date	1975-76?	1982
Equipment	clam	clam and scows
Disposal	confined disposal	open lake
Material	clay	sand
Quantity (m3)	4,600	27,161
Chemical data	VS-5.7%; OG-6.56; TP-0.41; COD-53.4; TKN-2.04; Hg-0.06; Pb-39.0; Zn-97.0	OG-0.983; TP-0.42; TKN-1.72; COD-38.4; PCB-0.002; Cd-1.4; Cu-9.0; Zn-30.6; Cr-12.2
Reference	Dredging Subcommittee (1982)	Sediment Work Group (1990)

Location	Hamilton Harbour (main channel & piers 12, 13)	Hamilton Harbour
Date	1978	1976
Equipment	clam and hydraulic	hydraulic
Disposal	confined in harbour	confined disposal facility
Material	silt, organic silt	organic silt, sand
Quantity (m3)	146,200	127,766
Chemical data	No chemical data	No chemical data
Reference	Dredging Subcommittee (1982)	Dredging Subcommittee (1982)

Appendix G. Continued.

Location	Hamilton Harbour (Emerald St. slip)	Hamilton Harbour (Wentworth St. slip)
Date	1980	1980
Equipment	hydraulic suction	hydraulic suction
Disposal	confined	confined
Material	unknown	unknown
Quantity (m3)	14,826	14,690
Chemical data	No chemical data	No chemcial data
Reference	Sediment Work Group	Sediment Work Group (1990)

Location	Hamilton Harbour (berth dredging)	Hamilton Harbour (channel dredging)
Date	1983	1983
Equipment	clam and scows	cutter suction
Disposal	confined	confined
Material	Class B sand, v. fine sand	Class B
Quantity (m3)	47,514	159,018
Chemical data	VS-9.0%; OG-7.9; TP-1.6; TKN-2.2; COD-121.6; PCB-7.6; Hg-0.5; Pb-244.2; As-7.0; Cd-4.08; Cu-70.0; Zn-1442.6; Cr-264.2; Ni-34.0	VS-21.4%; OG-7.97; TP-3.02; TKN-6.4; COD-215.5; PCB-1.7; Hg-0.63; Pb-169.0; As-9.8; Cd-2.45; Cu-107.2; Zn-1036.0; Cr-387.8; Ni-49.0
Reference	Sediment Work Group (1990)	Sediment Work Group (1990)

Appendix G. Continued.

Location	Hamilton Harbour (Burlington Ship Canal)	Grimbsy Harbour
Date	1984	1984
Equipment	dipper	clam excavator
Disposal	confined (& open)	re-used
Material	sand	sand, silt
Quantity (m3)	38,924	4,895
Chemical data	TP-0.644; TKN-0.274; PCB-0.048; Pb-8.3; As-2.3; Cd-1.0; Cu-11.6; Zn-99.3; Cr-13.0; Ni-22.0; COD-2.86	PCB-0.055; Hg-0.044; Pb-34.2; As-2.0; Cu-81.0; Zn-101.0; Cr-66.8; Ni-27.8
Reference	Sediment Work Group (1990)	Sediment Work Group (1990)

Location	Grimsby
Date	1985
Equipment	dragline/excavator
Disposal	beach
Material	unknown
Quantity (m3)	4,895
Chemical data	No chemical data
Reference	Sediment Work Group (1991)

Appendix G. Continued.

Location	Ogdensburg Harbor (St. Lawrence River)	Ogdensburg Harbor (St. Lawrence River)
Date	1977	1984
Equipment	hopper	no dredging
Disposal	confined	no dredging
Material	silt	sand
Quantity (m3)	26,703	
Chemical data	VS-2.04%; COD-196.6; OG-4.93; TKN-0.87; NH3-0.19; TP-0.17; PCB-7.0; Hg-0.40; Pb-88.0; As-7.0; Cd-2.9; Cu-50.0; Zn-866.0; Cr-37.0; Ni-39.0 Data averaged over entire harbor	VS-6.4%; OG-5.4; TP-0.79; TKN-1.7; NH3-0.048; COD-1.0; Hg-0.24; Pb-33.0; As-8.5; Cd-1.4; Cu-18.9; Zn-298.0; Cr-19.9
Reference	Dredging Subcommittee (1982)	Sediment Work Group (1990)

Location	Oswego Harbor (west basin & entrance)	Oswego Harbor (west basin, outer harbor)
Date	1975	1976
Equipment	hopper	hopper
Disposal	open lake	open lake
Material	silt	silt
Quantity (m3)	82,979	30,737
Chemical data	VS-3.95%; COD-47.6; OG-0.89; TKN-1.07; NH3-0.086; TP-0.81; Hg-0.10; Pb-49.0; As-9.0; Cd-1.4; Cu-36.0; Zn-110.0; Cr-17.0; Ni-17.0	VS-4.17%; COD-54.2; OG-0.82; TKN-1.25; NH3-0.096; TP-0.71; Hg-0.10; Pb-28.0; As-100.0; Cd-1.3; Cu-33.0; Zn-104.0; Cr-17.0; Ni-17.0
Reference	Dredging Subcommittee (1982)	Dredging Subcommittee (1982)

Appendix G. Continued.

Location	Oswego Harbor	Oswego Harbor
Date	1977	1979
Equipment	hopper	hopper
Disposal	open lake	open lake
Material	silt	silt
Quantity (m3)	53,803	21,398
Chemical data	VS-3.95%; COD-51.2; OG-0.92; TKN-1.22; NH3-0.104; TP-0.73; Hg-<0.01; Pb-31.0; As-9.0; Cd-1.6; Cu-32.0; Zn-117.0; Cr-11.0; Ni-18.0	VS-3.7%; COD-44.0; OG-0.85; TKN-0.74; NH3-0.09; TP-0.625; Hg-<0.1; Pb-22.0; As-9.0; Cd-1.1; Cu-33.0; Zn-99.0; Cr-16.0; Ni-18.0
Reference	Dredging Subcommittee (1982)	Dredging Subcommittee (1982)

Location	Oswego Harbor (harbour channel)	Oswego Harbor (harbour channel)
Date	1982	1984
Equipment	hopper	clamshell
Disposal	open lake	open lake
Material	silty sand	silty sand
Quantity (m3)	27,363	9,434
Chemical data	VS-2.6%; OG-0.6; TP-0.64; TKN-0.88; NH3-0.042; COD-36.0; Hg-0.096; Pb-37.2; As-1.76; Cd-0.73; Cu-22.8; Zn-63.2; Cr-11.7; Ni-17.4; Samples collected 1981	VS-2.6%; OG-0.6; TP-0.64; TKN-0.88; NH3-0.042; COD-36.0; Hg-0.096; Pb-37.2; As-1.76; Cd-0.73; Cu-22.8; Zn-63.2; Cr-11.7; Ni-17.4; Samples collected 1981
Reference	Sediment Work Group (1990)	Sediment Work Group (1990)

Appendix G. Continued.

Location	Oswego	Great Sodus Bay Harbor
Date	1987	1975
Equipment	trailing suction	hopper
Disposal	open lake	open lake
Material	unknown	sand
Quantity (m3)	34,255	8,278
Chemical data	No chemical data	VS-0.47%; COD-2.0; OG-1.12; TKN-2.27; NH3-0.206; TP-1.97
Reference	Sediment Work Group (1991)	Dredging Subcommittee (1982)

Location	Great Sodus	Little Sodus Harbor
Date	1981	1975
Equipment	no dredging	hopper
Disposal	no dredging	beach nourishment
Material	sand	sand
Quantity (m3)		1,612
Chemical data	VS-2.63%; OG-0.279; TP-0.527; TKN-0.653; NH3-0.036; COD-32.5; Hg-0.053; Pb-15.6; As-2.06; Cd-0.48; Cu-10.3; Zn-50.3; Cr-8.0; Ni-13.3	VS-0.35%; COD-5.4; OG-<0.001; TKN-0.23; Hg-<0.01; Pb-0.62; Zn-4.7
Reference	Sediment Work Group (1990)	Dredging Subcommittee (1982)

Appendix G. Continued.

Location	Little Sodus	Oak Orchard Creek
Date	1981	1984
Equipment	no dredging	no dredging
Disposal	no dredging	no dredging
Material	silty sand	sandy silt
Quantity (m3)		
Chemical data	VS-0.42%; OG-0.12; TP-0.36; TKN-0.095; NH3-0.005; COD-5.2; Hg-0.04; Pb-6.0; As-1.0; Cd-0.17; Cu-5.0; Zn-19.0; Cr-2.5; Ni-8.0	VS-3.33; OG-0.24; TP-0.66; TKN-1.44; NH3-0.058; COD-29.4; Hg-0.13; Pb-13.9; As-3.2; Cd-1.2; Cu-23.1; Zn-81.6; Cr-9.02; Ni-17.9; PCB (1221, 1232, 1242, 1248, 1254, 1260) - 0.10 each
Reference	Sediment Work Group (1990)	Sediment Work Group (1990)

Location	Oak Orchard	Irondequoit
Date	1985	1988
Equipment	bucket	bucket
Disposal	open lake	open lake
Material	silt, fine sand	unknown
Quantity (m3)	6,994	4,201
Chemical data	VS-3.6%; OG-0.1; TP-0.59; NH3-0.022; COD-14.6; Hg-0.072; Pb-11.8; As-4.0; Cd-1.08; Cu-14.0; Zn-69.6; Cr-8.6; Ni-13.2; PCB(1221, 1232, 1242, 1248, 1254, 1260) - 1.0 each congener	No chemical data
Reference	Sediment Work Group (1991)	Sediment Work Group (1991)

Appendix G. Continued.

Location	Rochester Harbor	Rochester Harbor
Date	1975	1976
Equipment	hopper	hopper
Disposal	open lake	open lake
Material	silt	silt
Quantity (m3)	200,539	94,046
Chemical data	VS-3.09%; COD-30.0; OG-0.71; TKN-0.81; NH3-0.078; TP-0.57; Hg-<0.1; Pb-31.0; As-9.0; Cd-3.7; Cu-25.0; Zn-120.0; Cr-20.0; Ni-20.0	VS-3.25%; COD-20.7; OG-0.6; TKN-0.43; NH3-0.066; TP-0.44; Hg-<0.1; Pb-33.0; As-9.0; Cd-2.8; Cu-24.0; Zn-107.0; Cr-19.0; Ni-19.0
Reference	Dredging Subcommittee (1982)	Dredging Subcommittee (1982)

Location	Rochester Harbor	Rochester Harbor
Date	1976	1977
Equipment	hopper	hopper
Disposal	open lake	open lake
Material	silt	silt
Quantity (m3)	132,569	103,601
Chemical data	VS-2.96; COD-33.4; OG-0.75; TKN-0.81; NH3-0.077; TP-0.62; Hg-<0.1; Pb-28.0; As-6.0; Cd-3.7; Cu-25.0; Zn-143.0; Cr-19.0; Ni-20.0	VS-3.37%; COD-36.0; OG-0.77; TKN-1.02; NH3-0.091; TP-0.68; Hg-<0.1; Pb-33.0; As-11.0; Cd-4.3; Cu-29.0; Zn-146.0; Cr-22.0; Ni-22.0
Reference	Dredging Subcommittee (1982)	Dredging Subcommittee (1982)

Appendix G. Continued.

Location	Rochester Harbor	Rochester Harbor
Date	1978	1979
Equipment	hopper	hopper
Disposal	open lake	open lake
Material	silt	silt
Quantity (m3)	246,502	216,155
Chemical data	VS-3.09; COD-30.0; OG-0.71; TKN-0.81; NH3-0.078; TP-0.57; Hg-<0.1; Pb-31.0; As-9.0; Cd-3.7; Cu-25.0; Zn-120.0; Cr-20.0; Ni-20.0	VS-3.07; COD-28.6; OG-0.69; TKN-0.67; NH3-0.073; TP-0.55; Hg-<0.1; Pb-30.0; As-7.0; Cd-3.4; Cu-25.0; Zn-130.0; Cr-19.0; Ni-20.0
Reference	Dredging Subcommittee (1982)	Dredging Subcommittee (1982)

Location	Rochester Harbor (harbor)	Rochester Harbor (harbor)
Date	1980	1981
Equipment	lyman	lyman
Disposal	open lake	open lake
Material	silty sand	silty sand
Quantity (m3)	113,676	82,096
Chemical data	OG-0.74; TP-0.71; COD-49.6; TKN-1.14; NH3-0.13; Hg-0.18; Pb-28.08; As-4.3; Cd-2.48; Cu-23.5; Zn-102.9; Cr-10.2; Ni-14.4 Samples collected in 1980	OG-0.74; TP-0.71; COD-49.6; TKN-1.14; NH3-0.13; Hg-0.18; Pb-28.08; As-4.3; Cd-2.48; Cu-23.5; Zn-102.9; Cr-10.2; Ni-14.4 Samples collected in 1980
Reference	Sediment Work Group (1990)	Sediment Work Group (1990)

Appendix G. Continued.

Location	Rochester Harbor (harbor channel)	Rochester Harbor (harbor channel)
Date	1982	1984
Equipment	hopper	clamshell
Disposal	open lake	open lake
Material	silty sand	silty sand
Quantity (m3)	108,897	227,973
Chemical data	OG-0.74; TP-0.71; COD-49.6; TKN-1.14; NH3-0.13; Hg-0.18; Pb-28.08; As-4.3; Cd-2.48; Cu-23.5; Zn-102.9; Cr-10.2; Ni-14.4 Samples collected in 1980	OG-0.74; TP-0.71; COD-49.6; TKN-1.14; NH3-0.13; Hg-0.18; Pb-28.08; As-4.3; Cd-2.48; Cu-23.5; Zn-102.9; Cr-10.2; Ni-14.4 Samples collected in 1980
Reference	Sediment Work Group (1990)	Sediment Work Group (1990)

Location	Rochester	Rochester
Date	1986	1987
Equipment	trailing suction	trailing suction
Disposal	open lake	open lake
Material	unknown	unknown
Quantity (m3)	314,107	114,562
Chemical data	VS-2.6%; OG-0.54; TP-0.52; TKN-0.69; NH3-0.025; COD-34.1; PCB(1242)-0.10; Pb-15.2; As-7.6; Cd-1.4; Cu-26.4; Zn-77.5; Cr-9.8; Ni-18.4 Sampled in 1985	Use 1985 chemical data
Reference	Sediment Work Group (1991)	Sediment Work Group (1991)

Appendix G. Continued.

Location	Rochester	Olcott
Date	1988	1981
Equipment	trailing suction	no dredging
Disposal	open lake	no dredging
Material	unknown	sandy silt
Quantity (m3)	138,051	
Chemical data	Use 1985 chemical data	OG-0.0; NH3-0.078; COD-0.7; PCB(1242)-0.31; Hg-0.13; Pb-53.8; As-5.8; Cd-0.8; Cu-40.1; Zn-175.6; Cr-28.8; Ni-50.9
Reference	Sediment Work Group (1991)	Sediment Work Group (1990)

Location	Olcott	Olcott
Date	1985	1989
Equipment	bucket	no dredging
Disposal	open lake	no dredging
Material	sand, silt	sand, silt
Quantity (m3)	3,926	
Chemical data	VS-3.6%; OG-0.64; TP-0.63; TKN-0.58; NH3-0.021; COD-8.04; Hg-0.70; Pb-65.7; As-4.4; Cd-45.2; Zn-270.6; Cr-28.1; Ni-33.6; PCB(1221, 1232, 1242, 1248, 1254, 1260) - 1.0 for each congener	TP-0.92; TKN-1.7; NH3-.054; COD-71.2; Hg-0.508; Pb-64.9; As-9.7; Cd-9.2; Cu-53.6; Zn-337.7; Cr-29.4; Ni-48.2; PCBs (all congeners) - 0.0
Reference	Sediment Work Group (1991)	Sediment Work Group (1991)

Appendix G. Continued.

Location	Wilson Harbor	Wilson
Date	1977	1981
Equipment	clam	no dredging
Disposal	open lake	no dredging
Material	sand	sand silt
Quantity (m3)	6,837	
Chemical data	VS-4.73%; COD-55.8; OG-0.626; TKN-1.56; Hg-0.03; Pb-15.6; Zn-42.7	OG-0.0; NH3-0.073; COD-0.6; PCB(1242)-0.37; Hg-0.26; Pb-25.5; As-8.4; Cd-0.54; Cu-26.2; Zn-59.3; Cr-23.8; Ni-14.2
Reference	Dredging Subcommittee (1982)	Sediment Work Group (1990)

Volatile solids (VS), oil and grease (OG), total phosphorus (TP), and COD in mg/g unless otherwise indicated. All metals and PCB data in ug/g.

Data sources: 1974-79 Dredging Subcommittee (1982)
1980-84 Sediment Work Group (1990)
1985-89 Sediment Work Group (1991)

APPENDIX H

SUMMARY OF STUDIES ON EUTROPHICATION IN LAKE ONTARIO

Reference	Leonardelli (1990)	Niagara River RAP Team (1990)
Location	Lake Ontario	Niagara River RAP (Ontario)
Time of Collection	1988	1970s and onwards
Species	NI	NI
Findings	high concentrations of P in detergents, reduction in P loads will be offset by growth of human population	turbidity problems in Welland River, P exceeds guidelines at mouth, P concentrations now decreasing, but N concentration increasing
Contaminant	P	P, N
Contaminant source	detergents, sewage	Welland River, Cyanamid Fort Erie and Niagara Falls STPs
Type of Report	Primary	Tertiary
Type of Evidence	Experimental	Observational

Reference	Metro Toronto RAP Team (1988)	Hamilton Harbour RAP Team (1989)
Location	Metro Toronto waterfront	Hamilton Harbour
Time of Collection	1960s and onwards	1960s and onwards
Species	Cladophora, phytoplankton	NI
Findings	P exceeds guidelines, Cladophora along western beaches, nutrients in sediments exceed guidelines, also clarity problems	P, ammonia and N exceed guidelines, DO below guidelines, algal blooms occur. Nutrients in sediments exceed guidelines
Contaminant	P	P, ammonia, N
Contaminant source	lakefilling, Humber & Ashbridges Bay STPs, CSO overflows	Hamilton STP, CSO overflows Stelco and Dofasco steel mills
Type of Report	Tertiary	Tertiary
Type of Evidence	Observational	Observational

Appendix H. Continued.

Reference	Bay of Quinte RAP Team (1990)	Port Hope RAP Team (1989)
Location	Bay of Quinte	Port Hope Harbour
Time of Collection	1960s and onwards	1970s and onwards
Species	NI	NI
Findings	P exceeds guidelines and there are clarity problems	P exceeds guidelines, but eutrophication not considered impaired use. Increased concentration of nitrate compared to Lake Ontario
Contaminant	P, N	P, N
Contaminant source	STPs, nonpoint sediment feedback	Cameco and nonpoint sources
Type of Report	Tertiary	Tertiary
Type of Evidence	Observational	Observational
Reference	Stevens and Neilson (1987)	Stevens (1988)
Location	Lake Ontario (OS)	Lake Ontario (NS and OS)
Time of Collection	1967-1982	1981-1982
Species	NI	NI
Findings	spring TP shows decreasing trend, less decrease in summer TP, no correlation between algal biomass and spring or summer TP. Demonstrates quick response time to reductions in P loading	reduction in P load, increase in N load. 50-60% reduction in Cladophora biomass
Contaminant	P	P, nitrate, nitrite
Contaminant source	40% from Niagara River	43% from Lake Erie, 30% from tributaries, and 18% from STPs
Type of Report	Primary	Tertiary
Type of Evidence	Observational	Observational

Appendix H. Continued.

Reference	Munawar et al. (1987)	Johannsson et al. (1985)
Location	Lake Ontario	Lake Ontario
Time of Collection	1970-1983	1981-1982
Species	NI - phytoplankton in general	NI - phytoplankton in general
Findings	decrease in phytoplankton biomass from 1970 to 1978 correlated with differences between 1972 and 1983 small plankton more affected by contaminants	decrease in phytoplankton biomass, decrease in P concentration, algal species shift to meso-trophic
Contaminant	P, metals	P
Contaminant source	NI	NI
Type of Report	Primary	Primary
Type of Evidence	Observational	Observational

Reference	Makarewicz (1985)	General Codes;
Location	Rochester, Oswego, Niagara R (NY)	NI - not indicated in paper
Time of Collection	1981	NS - nearshore location
Species	NI - phytoplankton in general	OS - offshore location
Findings	decrease in nutrient levels since 1974, localized areas of high nutrient loadings, spring open-lake SRP ranges from 3 to 29 ug/L	
Contaminant	P, N, metals	
Contaminant source	NI - various	
Type of Report	Primary	
Type of Evidence	Observational	